Atomic Energy Control Board
P.O. Box 1046
Ottawa, Canada
K1P 5S9

Commission de contrôle de l'énergie atomique
C.P. 1046
Ottawa, Canada
K1P 5S9

INFO-0070

A QUALITATIVE EVALUATION OF LONG-TERM PROCESSES GOVERNING THE BEHAVIOUR OF URANIUM MILL TAILINGS PLACED IN DEEP LAKES

A Study conducted for the

ATOMIC ENERGY CONTROL BOARD

by

BEAK CONSULTANTS LIMITED
TORONTO, ONTARIO

March 1982

RESEARCH REPORT

ABSTRACT

A Qualitative evaluation of long-term processes governing uranium mill tailings placed in lake sediments.

The emplacement of uranium mill tailings in deep lakes may provide an acceptable method of tailings disposal at certain sites in Canada. Disposal in a depositional environment typical of deep lakes appears to offer greater prospects for long-term stability than present methods of land-based management. From the regulatory point of view, it is necessary to know which factors should be taken into account in assessing the acceptability of such an approach. This report examines the environmental variables governing the behaviour of radionuclides and trace elements in the groundwater systems, lake water, and finally in the biosphere over the short and long-term. Physical, chemical and biological factors are each considered. Conclusions are presented in terms of points for and against disposal in deep lakes.

The opinions expressed in this report are those of the authors and do not necessarily reflect those of the Atomic Energy Control Board.

FOREWARD

This document discusses and summarizes the most important points in an annex document, bearing the same title, submitted by BEAK CONSULTANTS LIMITED to the Atomic Energy Control Board of Canada in December 1981. The annex should be consulted for literature references, tables, and detailed explanations. For convenience, citations for relevant sections of the annex are given in this document.

INTRODUCTION

One of the key environmental questions to be answered in the development of nuclear energy concerns the long-term management of uranium mine tailings. Management must be as secure as possible because uranium tailings contain potentially toxic substances, including not only residual uranium and its decay products, but also other associated radioactive or chemically toxic metals originally found in ore bodies. Typical substances of concern include radioisotopes of the uranium (e.g., U-238, Th-230, Ra-226) and thorium (e.g., Th-232) decay series, and heavy metals (e.g., As, Pb, Fe, Mn, Ni). The commonly used method of valley dam management has several fundamental containment limitations. Erosion of the tailings mass may lead to wind-driven atmospheric transport and surface runoff of chemically and radiotoxic material. Percolation of surface water down through the tailings can leach contaminants and carry them into groundwater systems and under certain circumstances into contact with the biosphere. Bacterial attack on pyrite minerals in unsaturated zones causes substantial acid generation and increased contaminant mobilization. Plant material growing on the tailings mass can mobilize elements through root uptake. These elements can then enter food chains when those plants are grazed.

Because of these concerns, another option is now under consideration for several sites in Canada; that of deep-lake disposal. A tailings mass situated at the bottom of a deep lake is in a depositional, rather than erosional, environment because of the continuous sedimentation of organic and inorganic matter produced in the lake watershed and because of the protection against erosion afforded by the overlying waters of the lake itself. Because of continual sedimentation, the layer between the tailings and the overlying waters becomes thicker with time. Assuming that the lake is deep enough to place the disposal site well below the photosynthetic compensation depth (below which there can be no plant growth due to lack of light), root uptake by aquatic macrophytes is not a problem. Another advantage in deep-lake disposal is that tailings so placed are buffered against perturbation from meteorological extremes such as hurricanes, floods, and droughts by a permanent layer of water. Deep lake basins may be relatively isolated from shallow local groundwater networks, so the near-surface flow is into and out of the upper waters of the lake. Finally, deep lake sediments tend to be anoxic, providing conditions that promote the geochemical stability of many of the trace elements found within the tailings. For these reasons, deep lake sediments are

generally viewed as sinks rather than sources for materials moving within a watershed. That is, they are zones for which little mass is mobilized and carried to downstream receiving waters relative to the total mass present.

Against these advantages, there are objections that must be considered. Since most lakes undergo complete vertical mixing at least once a year, any contaminants that do migrate from the tailings into the deep overlying waters, could be circulated upwards to the trophogenic zone and thence to aquatic and terrestrial food chains. The relative importance of this mechanism will depend upon leaching rates, direction and magnitude of physical transport, and possible long-term perturbation effects. There is a second objection if there is a potential migration pathway to deep groundwater systems. Depending upon site-specific considerations this may be detrimental or beneficial (e.g., transport to surface water systems as opposed to deep bedrock attenuation of contaminants).

In order to make a decision about any disposal scheme, all probable migration mechanisms and pathways must be qualitatively and quantitatively (where possible) examined and the resulting information weighed against similar data for alternative schemes. This document and the accompanying report provide a general guide for examining deep-lake disposal proposals by describing a conceptual framework and discussing specific variables as they fit into it.

THE APPROACH

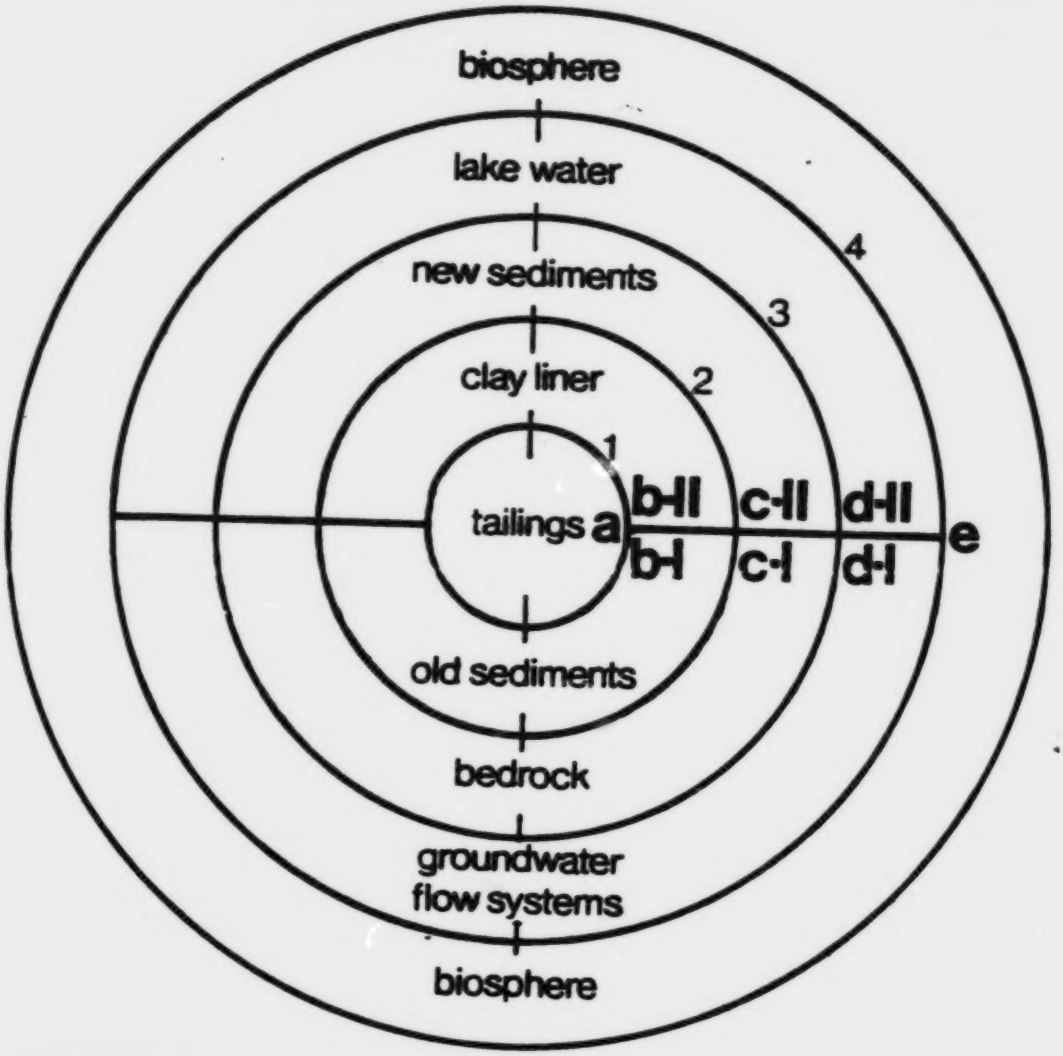
We have chosen to approach the problem in a "micro-to-macro" fashion, i.e., starting by considering the characteristics of material in the tailings mass and working outward, past various barriers, to the biosphere as a whole. This can be viewed conceptually as a series of concentric circles (Figure 1). The tailings mass is at the centre. Each band (A-E) represents the location of a migrating contaminant at a given stage in its progress, subject to various physical, chemical or biological processes, and each circle (1-4) represents a barrier, or interface, between one location and the next. A horizontal line divides the diagram into two halves, representing the two possible directions of migration, downwards into the groundwater-bedrock system (bottom half) or upwards into the overlying water (top half). The two halves reunite in the outermost band, the biosphere as a whole.

At each band or interface an evaluation of the suggested variables will give a prediction as to whether contaminant movement will be outward (away from the tailings), inward, or negligible.

Two time-scales have been considered. For the period of 0 to 100 years after placement of the tailings, short-term processes (generally irreversible) are most important, because the final geochemical state of the tailings may still be evolving, as physical consolidation may not be complete and long-term biochemical boundaries may still be developing. For the 100- to 10 000-year time-frame, processes such as cyclic changes in weather, seismic events, rare meteorological events, and slow diffusion systems becomes increasingly important and only the most general predictions can be made. The period beyond 10 000 years was assumed to be imponderable.

Figure 1

Conceptual schematic of major barriers between tailings and biosphere



Interfaces

1 tailings mass pore water
2 tailings barrier pore water
3 sediment-lake water inteface
4 aquatic food chain

T-3802C

A THE TAILINGS MASS (Section 1.1 of annex)

The tailings mass is the most important area of the diagram in Figure 1. Only if a contaminant escapes area A will the subsequent interfaces and bands be of concern. Furthermore, contaminants in the tailing mass are in a relatively confined area and thus more amenable to control and to predictive modelling. In the same manner, each successive band or circle can be seen as more critical than the ones beyond it.

Once placed at the lake bottom, the tailings mass will consist of particulate matter and interstitial pore water. The physical stability of the deep-lake environment permits us to assume that particulate matter will not be mobile as long as the final tailings mass lies completely within the depositional zone of the lake and is remote from any significant erosional forces. The first step in predicting migration of toxic elements is a knowledge of structure and content of the tailings mass. For example, one type of tailings might include: i) the leached host minerals (leached and milled ore); ii) materials sorbed or precipitated on the tailings minerals, and iii) chemical sludges or precipitates. These three tailing components have different contaminant concentrations and different leaching characteristics.

The next requirement for predictions of mobility is a knowledge of the leachability of contaminants from particulate forms into the pore water. Leachability will be controlled by the surface area of particles exposed to pore water and by geochemical and biological processes controlling the partitioning of contaminants between the soluble and particulate forms. Since the geochemical and biological-chemical properties may differ from one element to another, each potential contaminant must be considered individually.

Geochemical variables influencing partitioning of elements include:

1) pH - controls ion formation states and dissolution coefficients;
2) Eh (reduction-oxidation or redox potential) - determines whether oxidized or reduced forms of ions will occur;
3) ionic strength - influences the chemical activity of dissolved ions, and thus their ability to form precipitates;
4) buffering system - controls pH; and,
5) crystalline forms of minerals - may dissolve to change porewater concentrations, and may precipitate if supersaturated.

Important biochemical variables are:

1) carbon source and trace nutrients availability - necessary for bacterial communities;
2) microbial communities - influence redox potential and pH; and
3) chemical leaching agents - can desorb metals and may be produced by bacteria and higher plants.

The actual long-term pH, Eh, and ionic strength within the tailings mass is a function of i) the relative rates of dissolution of various minerals in the host tailings (e.g., an arsenide mineral), and chemical precipitates formed in the mill (e.g., gypsum, jarosite, calcite), ii) the chemical species entering the tailings mass with groundwater, and iii) whether or not particular microbial communities develop in the tailings mass. For example, the pH may decrease due to the development of a sulfur-oxidizing community when all calcite has been dissolved and transported from the tailings mass. At low pHs increasing amounts of gypsum will dissolve, increasing the ionic strength. Hence, while variables such as pH and Eh are master variables, they are determined and controlled by the particular geochemical and biochemical system which dominates at a particular point in time.

Temperature is what may be termed a master variable for both geochemical and biochemical processes because it controls inorganic and organic reaction rates.

Depending upon what combinations of values for these and other variables occur, a number of quite different scenarios can be predicted. For example, in a tailings mass containing no organics, both sulfide (S^{-2}) and a potential contaminant metal ion (M^{+2}) might be leached into the pore water and escape the tailings mass. However, if sufficient reduced iron (Fe^{+2}) were present (as the result of organic reactions), the iron might coprecipitate with the sulfide and the potential contaminant to reduce the loss rate from the tailings. In the latter case, the interplay of geochemical (Fe present) and biochemical (organic matter, bacteria) variables changes the partitioning.

Once the biogeochemical state of tailings has been defined, distribution coefficients between particles and pore water can be estimated. The next step is then to examine transport through the pore water.

A-I PORE WATER TRANSPORT

The second most important zone of Figure 1 is Circle 1, the transport processes through the pore water. These are the processes by which contaminants in solution may reach zone B of Figure 1 (upward or downward, depending upon transport gradients). On a short-term basis the tailings can be viewed as a "black box" with water flowing through. Once the particle/pore water distribution coefficient is established and if the pore water concentration is constant at a particular point in the tailings, transport will be controlled by two physical mechanisms: i) advective transport - water movement in one direction, (controlled by the hydraulic gradient in the tailings mass and the permeability of the medium); and, ii) diffusional - type processes - limited by concentration gradients and driven by the process in molecular physics of random collision with water molecules, or dispersion which is driven by molecular diffusion, advection and turbulence. Variables to be evaluated include directional permeability coefficients, hydraulic gradients, concentration gradients between tailings porewater and overlying and underlying strata, and the bulk diffusivity coefficient.

Long-term processes affecting physical transport include consolidation of the tailings mass (changing permeability and bulk diffusivity), shifts in hydraulic gradients caused by long-term geological processes or possibly by human perturbations, shifts in chemical gradients and response of the tailings mass to sudden accelerations caused by seismic or meteorological events.

B-I EXISTING LAKE SEDIMENTS (Section 1.3 of annex)

Concepts describing the role of lake sediments in governing the fate of contaminants in tailings, are discussed later (Section C-II, New Sediments). Important differences between existing (pre-disposal) lake sediments underlying the tailing mass and new sediments which cover the tailings include:

1) if general watershed weathering variables remain constant, the existing sediments will tend to be more consolidated and have reduced permeability and bulk diffusivity coefficients in comparison with post-disposal sediments; their values will be decreased by the additional weight of overlying tailings;
2) the existing sediments tend to contain older and more refractory organic matter and thus may be expected to have a lower potential for supporting microbial activity; and
3) the existing sediments would be physically isolated from the overlying water and thus not subject to physical disturbance (bioturbation) by aquatic organisms.

B-II TAILINGS BARRIER (Section 1.2 of annex)

A tailings barrier, installed on top of the original lake sediment or overlying the deposited tailings, represents a physical and a chemical barrier against both the mobilization of contaminants within the tailing mass and movement of contaminants from the tailing mass. The ability of the barrier to retard transport, both by geochemical retardation and restriction of advective flow is therefore of prime concern. As in the tailings mass itself, chemical adsorptive and dissolution processes and phyical advective and diffusive transport are involved. Possible barriers include liners composed exclusively or partially of clays.

The chemical composition of the barrier will determine what types of ions (e.g., H^+ or S^{-2}) and surfaces a soluble contaminant will encounter. Clay particles tend to have good adsorption properties for some radionuclides and heavy metals while a fine-grained collophane apatite is even better. A clay- apatite mixture shows promise as an excellent adsorber for radium.

The chemical composition of the barrier pore water is also important. Under anoxic conditions, metal sulfide precipitation can be a further barrier to migration. The presence of elements capable of forming other precipitates with anticipated contaminants could be deliberately enhanced within the barrier as a further control measure.

As in the tailings mass itself, pore water transport in the tailings barrier is controlled by permeability, diffusivity, and the magnitude and direction of hydraulic and chemical gradients. Molecular diffusion will depend on ionic concentration gradients between the barrier pore water and the interstitial water of the overlying sediments. The permeability and bulk diffusivity will be influenced by the amount of consolidation of the barrier. Consolidation is a long-term process which results in a re-packing of particles into void spaces. It is caused by the compacting load of the tailings mass itself. Clay compaction is strongly affected by the surface charges on the clay particles. A barrier composed of particles of uniform charge will not compact as well as a barrier of alternating lattices of positive and negative charges. The thickness of the liner will be a master variable determining contaminant passage time for a given set of physical and chemical conditions.

The stability of the barrier is also a concern. A short-term variable to evaluate is the cohesiveness of the liner. Before new sediment has had time to accumulate, the liner may be exposed to shear stresses from bottom currents, especially if weather extremes occur during this phase, or from gravitational forces if the barrier is not horizontal. Rapid settling and reconsolidation of the tailings mass after such physical disturbances may promote relatively minor increases in contaminant transport relative to long-term, constant transport processes. Over the long-term, resistance to perturbation from seismic events must be considered.

C-I BEDROCK (Section 3.0 of annex)

If leached contaminants leave the tailings mass and migrate through underlying lake sediments and glacial debris, they may eventually reach the bedrock (zone C-I, in Figure 1). The behaviour of contaminants in a bedrock environment is principally controlled by its physical structure, chemical nature and stability. The physical structure will determine the relative importance of porous media flow, flow along fractures, and diffusion as contributors to total contaminant transport. Important variables include bedrock type, matrix permeability, diffusivity, and the size, presence, density and continuity of fractures. As with the tailings barrier, the chemical nature of the bedrock fracture lining and rock matrix will determine its adsorptive capacity. If bulk permeability and diffusivity coefficients are low and adsorptive capacity is sufficiently high, then the bedrock can be viewed as a long-term geological sink for many radionuclides and other contaminants.

In the long term, the physical and chemical stability of the bedrock is critical to the security of the system. Instability would lead principally to more rapid transport through groundwater systems, although the physical collapse of the containment system and possibly the tailings mass itself cannot be ruled out in certain formations.

Variables to evaluate for physical stability include:

1) bedrock type - sedimentary, metamorphic or igneous;
2) past history of seismic activity;
3) presence and pattern of fractures - guide to recent geological history and stability;
4) long-term stability of fracture characteristics - reaction to future stress changes; and
5) rate of post-glacial isostatic rebound and plate-tectonic movements - their relationship to observed bedrock fracturing, vertical fracture and fracture aperture distribution.

Chemical reactivity can be evaluated by studying:

1) the nature of fracture surface coatings and their potential for geochemically retarding the effective rate of contaminant migration;
2) the mineralogy of the bulk rock matrix and its potential for retarding contaminants by adsorbing contaminants which diffuse into the rock matrix;
3) the potential for interaction of materials migrating from the tailing mass with fracture surfaces. An example of this is an acidic tailings leachate migrating along carbonate-coated fractures. This will cause dissolution of carbonates, and the resultant increase in fracture aperture will alter the rate of contaminant transport.

C-II NEW SEDIMENTS (Section 1.3, 2.2 and 4.0 of annex)

Whether an engineered tailings liner exists or not, naturally sedimenting particulate matter from the lake and surrounding watershed provides an additional barrier between the tailings mass and the lake water. Although physical and biological translocation of particulate matter is a possibility, especially in shallow water sediments, it is nevertheless reasonable to assume that, over the long term, in deep lakes, most contaminant movement occurs in soluble forms. For these soluble constituents, chemical equilibria and physical transport play key roles.

As in the tailings mass, vertical advection and diffusion are the principal transport mechanisms. Advection will be influenced by hydraulic pressures (groundwater flow and squeezing out of pore water due to consolidation). Mineralogy, grain size distribution, porosity, density and organic matter content are variables of interest to estimate physical and chemical transport potential. The density and community composition of benthic invertebrates will control biologically-induced water movements in the upper sediments. Diffusion-related variables include chemical gradients, adsorption and porosity; these variables can be formulated into an effective diffusion coefficient for the sediment. Chemical gradients of other elements could be significant in regulating contaminant movement if they influence ionic strength and compete for adsorption sites. The consolidation characteristics of the new sediments accumulating above the tailings will influence both advection and diffusion, as will long-term stability. Key variables in determining the consolidation characteristics are the sediment's shear stress resistance, macroinvertebrate densities, and flocculant character of the settling particles.

Biogeochemical process within the new sediments will determine the concentrations of soluble elements available for physical transport across this growing barrier. Critical geochemical processes include:

1. deposition of heavy metals, mostly as inert inorganic compounds;
2. precipitation as hydroxides, carbonates, sulphides and phosphates, particularly in alkaline waters;
3. sorption and cation exchange onto particle surfaces (clays, Fe and Mn-oxides, humics, etc.);
4. complexing or chelating of dissolved metals (humic and fulvic acids) by dissolved organics;
5. co-precipitation of metals with hydroxides, carbonates and sulfides.

The relative rates and magnitudes of these processes must be determined in order to make predictions. Essential variables for such calculations are:

A) Sediment properties:

1. particle size distribution - controls permeability, porosity and surface area available for reaction with solutes;
2. particle surface chemistry - will determine adsorption isotherms and cation exchange capacity and hence, the resultant sediment attenuation capacity.

B) Pore water chemistry

1. pH - affects mineral breakdown, adsorption, complexing capacity, solubility and precipitation rates;
2. buffering capacity - controls pH, precipitation;
3. Eh (redox potential) - controls reaction rates and direction, speciation of metals;
4. major ion concentrations - influence biochemical reactions, and possible precipitates;
5. contaminant concentrations - a monitoring tool and controls diffusion gradients.

Biochemical processes also control particle-pore water distribution coefficients for contaminants. Besides influencing sediment cation-exchange capacity, the abundance of organic matter also determines what microbial communities will exist in the sediments. These biological systems, in turn, determine what the redox potential, one of the most important chemical variables, will be. The oxygen demand of sediments can decrease oxygen concentrations to the point where the redox potential drops, causing some metal oxides, with their coprecipitated metals, to become soluble. Opposing this tendency is the capacity for sulphides to form metal precipitates at negative redox potentials. A neutral or negative redox potential is anticipated for most sediments.

Another series of mechanisms, but to date poorly understood, involves the complexing capacities of various organic compounds (humics, amino acids, etc.) which can compete with inorganic particulates for dissolved metals and thus increase or decrease their solubility. Direct metal uptake by microbes and bioturbation by macroinvertebrates adds further to the complexity of the sediment-pore water-biota system. Finally, the stratification of biological communities, chemical variables and physical structure makes the sediments a multi-layered barrier and thus difficult to treat as a "black box".

Given all of these complex physical, chemical and biological mechanisms, the thickness of the sediment layer stands out as one of the major controlling variables. Since the sediments accumulate over the tailing mass with time, the sedimentation rate is the controlling factor. The erosional characteristics of the lake's watershed (inorganic and allocthonous organic matter inputs), and the productivity of the lake itself (autochthonous organic matter) are of primary interest in predicting sedimentation rates.

D-I DEEP GROUND WATER (Sections 3.0 and 4.0 of annex)

The deep groundwater system is critical because, as the next-to-last band (zone D-I) in Figure 1, contaminants moving from it have the potential to impinge directly on the biosphere. Thus, the most important questions to ask are: i) in what manner does the deep groundwater system underlying the tailings interconnect with surface waters; and ii) what are the factors within that flow system which control a) the rate of water movement, b) the degree of contaminant attenuation, and c) the rate of contaminant dilution and dispersion within the flow system. To answer these questions requires a knowledge of regional groundwater patterns and estimates of fracture and matrix permeability, hydraulic pressures, diffusion coefficients, and the adsorption capacity of the medium. After balancing transport time against adsorption during transport, a final rate of contaminant discharge, or "bleed rate" can be estimated and compared with acceptable levels of contaminant release. The stability of the groundwater system is a long-term master variable, and is a function of the bedrock stability (see Section C-I) long-term weather cycles influencing the water table (Section 6.0 of annex), and human induced perturbations (Section 7.0 of annex).

D-II LAKE WATER (Sections 2.0, 6.3 and 6.4 of annex)

The lake water body is two steps removed from the centre of Figure 1. It is the last line of defence before entry into the open environment, and accordingly, interfaces physically and biologically with the biosphere. Hence, while viewed as a defence for the open environment, contaminant concentrations in it are a direct concern due to water contact activities of humans.

For a given rate of transport of a contaminant from the tailings mass into the lake water, a number of mechanisms can operate to mediate biological availability. At the sediment-water interface, redox potential can

determine whether an element will migrate into the water. Strong bottom currents can increase the rate of contaminant loss by disturbing the sediments and by increasing the chemical gradients across the interface. Of more fundamental interest is the vertical physical transport system within the lake. Advective transport is a function of vertical water circulation. Most lakes in temperate-latitudes are stratified by thermally-induced density gradients for most of the year, but undergo complete vertical mixing (mixis or turnover) at least once (monomictic lakes) and usually twice (dimictic) annually. Typically, fall turnover occurs when summer stratification (lighter warmer water near the surface and colder, denser water down to 4°C near the bottom; called orthograde stratification) breaks down. This occurs when cooler air temperatures cause surface waters to cool and become close to those of bottom waters. Winter stratification is retrograde (lighter water near 0°C at the surface, denser water near 4°C at the bottom) and breaks down as warm spring air temperatures heat up the surface waters.

As previously indicated (see Introduction), periods of turnover are times when advective vertical transport of metals which have bled out of the sediments from bottom waters to surface waters would occur. Possible measurements of lake circulation characteristics are:

1. frequency and duration of turnovers - number of days per year when bottom and surface waters are mixing;
2. maximum depth of vertical overturn - is mixing complete to the bottom of the deepest basin;
3. rate of thermocline (density gradient) erosion during summer stratification - susceptibility to wind-driven mixing, measured by vertical temperature profiles;
4. Stability of stratification (e.g., Richardson Number) - resistance to wind-induced shear stress.

In addition to thermal density gradients, salinity gradients also contribute to vertical stability in some lakes (see Section 7.2, main report). If a lake is sufficiently deep and its bottom waters are more saline than its surface waters, wind-induced shear stress may not be able to overcome vertical density differences even when temperatures are vertically isothermal. Such lakes are called meromictic. Sources of salinity include seawater incursions into coastal lakes during storms (ectogenic), dissolution from geologic deposits (crenogenic), and decomposition of organic matter (biogenic). In as much as meromixis precludes advective transport to surface waters and as more saline water reduces the diffusivity coefficient, this condition is desirable in a tailings disposal system. In some cases (e.g., Garrow Lake, NWT), naturally occuring meromictic lakes may be close to mining sites, making them potential disposal sites. To date, only a limited number of meromictic lakes have been studied in Canada, but there are undoubtedly many more still undiscovered. In looking for meromictic lakes, variables to be measured include:

1. vertical salinity (Total Dissolved Solids) gradient: (picnocline)
2. year-round thermal stratification
3. adiabiatic heating of bottom waters
4. deep-water blooms of sulfur-reducing bacteria
5. relative depth - exposure to wind vs. vertical distance to be mixed.

Another option is to attempt to artificially induce meromixis by adding salts to bottom waters. Calculation of the salinity gradient required to produce a given degree of vertical stability is fairly straight forward, but meromixis is produced only by the proper combination of salinity, lake relative depth, and protection from wind-induced shear stress by hills. Wind-driven mixing is difficult to predict because of our lack of knowledge of atmospheric boundary-layer effects when wind passes over a lake surface, and because of the uncertainty about the frequency of storms with extremely high winds. Specific factors include the presence of surrounding hills, length and width of the lake (fetch) and prevailing wind directions. If a management option is to involve the creation of an artificial chemocline within a lake, care must be taken to ensure the stability of the induced meromixis. Artificial eutrophication of the lake and hence, the possible creation of a long-term biogenically stabilized meromixis is one such possibility.

Vertical diffusion is also critical to upward transport. Mathematical models for predicting diffusion as a function of temperature gradients are currently being developed. Eddy diffusion coefficients tend to be higher where thermal stability is low.

In predicting the net upward movement of dissolved contaminants bleeding from the sediments, the chemical and biological factors controlling their particulate/solute distribution coefficients in the water column are of great importance. Metal ions that become particulate tend to settle back down into the sediments, while soluble ions are subject to upward physical transport and potential biological uptake. Important mineralization and precipitation mechanisms are:

1. ionic precipitation - when the dissolved concentration of a metal exceeds its solubility, or when metal ions are adsorbed onto inorganic particles;
2. biogenic uptake - uptake of metals by organisms, mostly phytoplankton, which then settle to the sediments; and,
3. adsorption by humic compounds - ion scavenging by large organic molecules. This is especially important in Canadian Shield dystrophic lakes.

Chemical variables useful in predicting adsorption and distribution coefficients and the probability of ionic precipitation are:

1. pH - controls solubility; it depends in many lakes upon the bicarbonate buffering system. In other lakes it may be controlled by other minerals or organics.
2. oxygen regimes - control microbial communities at sediment-water interface; these communities control redox potentials;
3. organic carbon, particulate (POC) and dissolved (DOC) - potential substrates for bacterial growth, and also surfaces for chemical adsorption and subsequent settling; they influence oxygen consumption;
4. major cations and anions - potential nutrients for biological growth; provide evidence of allochthonous inorganic inputs;
5. background contaminant levels - an indication of how significant bleed rates from tailings will be in terms of original conditions; and
6. biological productivity factors (N, P, Si, essential trace metals, vitamins, organic compounds) - potential for biogenic precipitation and meromix induction and/or maintenance.

Biological variables needed for predicting net upward transport include:

1. bacterial community structure - aerobic or anaerobic;
2. phytoplankton - capacity to adsorb and take up metals;
3. zooplankton - capacity to concentrate and settle out metals to allow their entry into food chains and to regenerate nutrients for phytoplankton growth;
4. benthic macroinvertebrates - bioturbation of upper sediments; and
5. fish - capacity to recycle nutrients, to crop zooplankton and benthos and to provide an important food-chain link to man.

In a final analysis of contaminant mobilization, the bleed rate from tailings through sediments is used as a source term, and upward transport mechanisms are balanced against rates of mineralization and sedimentation to predict loss rates to the surface waters. Horizontal circulation can lead to transport to lake outlets or to shallower areas.

E THE BIOSPHERE (Sections 1.3, 2.1, 5.0 and 7.0 of annex)

The ultimate concern in any contaminant disposal problem is that humans and organisms that they value, whether for nutritional, aesthetic, or ecological reasons must be protected.

The first organisms to incorporate contaminants lost from a deep-lake tailings mass would be microscopic plankters. These organisms are not in themselves highly valued, but they are the food sources of trophic pathways culminating in higher organisms which often enter the human diet. The problem is compounded by the tendency for some trace elements in some biological systems to become more and more concentrated at progressively

higher trophic levels. A variable used to evaluate this concentration process is the bioaccumulation factor (BF), the ratio of organisms' concentration to water concentration of an element. If surface water concentrations can be predicted, then BF values can be used to estimate flesh concentrations and resultant chemical or radiological toxicity. A difficulty in Canada is that most BF values available in the literature were observed from calcareous lake systems, while a great many Canadian lakes are on the Precambrian Shield, where water chemistry and biota, and thus contaminant behaviour, are quite different. Application of BF values from hard waters to these lakes is probably not valid.

In addition to bioaccumulation in the disposal lake itself, loss of contaminants such as metals through the lake outlet to downstream waters or to deep groundwater systems are two other pathways for entry into the biosphere. Their relative importance depends upon with which biota these pathways intersect.

Effects of the biosphere upon the lake must also be considered (Section 5.0 of annex). Of primary concern in this regard is the export of materials from the drainage basin to the lake itself. In most lakes, the main source of nutrients (for phytoplankton growth and biogenic precipitation of contaminants), mineral salts (buffering capacity, chemical precipitation), and humic material (complexation, chelation of metal elements) is the watershed. Furthermore, a large number of Canadian lakes are in the dystrophic category, in which most of the organic matter in a lake is of an allochthonous origin. In these lakes, then, the driving force for bacterial communities is energy fixed in the watershed and transported to the lake by streams, surficial runoff and groundwater systems. Finally, watershed export will largely control the background levels of radionuclides against which bleed rates from the tailings mass must be compared.

For these reasons, estimates of material input to the lake are necessary. These may be expressed in units of mass/time, mass/surface area/time, or calculated from flow regimes and background concentrations. Short-term variables involved in such calculations include rock and soil composition, solubility and leachability, terrestrial productivity, land use, rainfall volume and chemistry, basin slope, and erosional characteristics. Predictions of long-term changes in export depend upon variables such as future land use patterns, seismic activity, meteorological cycles and extremes and hydrologic changes (especially man-made; Sections 6.1 - 6.4 and 7.1 of annex).

Beyond the fairly well definable hydrologic units of the lake and the drainage basin, it becomes very difficult to evaluate the relative importance of the major atmospheric variables and the ways in which they influence the lake and its watershed and the biosphere. The lake-drainage basin system is influenced by such atmospheric phenomena (Section 6.0 of annex) as solar

radiation, winds, precipitation (including long-range contaminant transport and anthropogenic acidification), temperature cycles and gaseous composition. All of these are subject to very long-term patterns. Finally, the effect that humans may have on individual lakes, drainage basins, and the biosphere as a whole are very difficult to predict. Short-term variables include land use, recreational fishing and hunting, and changing public perceptions of what is environmentally acceptable. Among long-range possibilities are changes in habitation patterns, atmospheric changes due to human activities (CO_2 cycle), forest fires, logging, lake eutrophication, acidification and hydrologic manipulations (dams, reservoirs, etc.).

The question of where the effects of the biosphere on the tailings mass (and vice-versa) becomes too diffuse and intractable to predict is one that must be decided by the regulatory agency responsible for a particular tailings management decision. For example, in a relatively small lake subject to intense recreational use, turbulence caused by outboard motors must be considered as a possible contributor to upward advective transport of tailings contaminants and to shear stress upon a newly-placed tailings barrier. For a lake of similar size and shape but in a more inaccessible region, this type of perturbation would probably be negligible. But long-range forecasts must take into consideration the possible effect of future population changes and possibly future technological developments.

CONCLUSIONS

This review of many of the mechanisms and processes affecting the mobilization and distribution of contaminants from uranium mine tailings placed in the bottom of a deep lake allows a series of statements to be made and conclusions to be drawn.

1. In a long-term geological context and in order to minimize the rate of contaminant dispersal, it is better to place mine tailings in local or regional depressions which are sedimentary environments rather than on topographic "highs" which are erosional environments.

2. Within the major uranium mining areas of Canada, the relatively high rainfall, thin surficial cover, low bedrock permeability and topography result in such depressions being naturally occurring lake basins. Other depressions such as open pits are man-made lake basins. They rapidly flood and become lakes upon the cessation of dewatering activities.

3. Accordingly, in all but the southern semiarid areas of Canada, below-grade disposal in sedimentary environments means disposal within lake sediments.

4. Disposal within lake sediments is a promising disposal option which may find application in Canada. It offers many advantages while at the same time having inherent drawbacks.
5. Deep-lake disposal is a passive disposal concept in theory requiring and indeed offering very little opportunity for future modification.

The advantages and disadvantages of deep lake disposal may be summarized briefly as follows. Most relate to containment versus dispersal approaches.

Advantages	Disadvantages
Placement in a depositional environment promotes increasing isolation with time.	Increasing isolation with time means they may be more difficult to recover for any reason at a future date.
A covering of water over the tailings reduces radon release to negligible levels and buffers the tailings against meteorological extremes.	A covering of water may allow rapid transport of contaminants from the tailings to surface waters and to downstream waters if they reach surface waters during periods of high runoff and/or lake turnover.
A covering of water and isolation of tailings at depth ensures a minimum of bioperturbation from humans and other natural communities.	During placement of tailings in a deep-lake basin, there is the opportunity for contaminant transport out of the system.
Seepage of contaminants from the tailings into deep flow systems underlying the lake basin offers the opportunity for geological attenuation at depth and effective isolation of the biosphere.	Determination of deep bedrock flow systems and geochemical attenuation capacity is difficult and expensive. The uncertainty involved with these aspects confounds the uncertainties associated with other variables such as surface water hydrology, lake circulation patterns sedimentation rates, etc.
Insertion of tailings into a reducing environment (of negative redox potential) places them in a situation in which most minerals (e.g., pyrites, arsenides) are stable. New minerals may precipitate, providing a mechanism for coprecipitating contaminants and new surfaces for contaminant adsorption.	Insertion of tailings into a reducing environment may destabilize precipitates (e.g., gypsum, jarosite, hydroxides of thorium and calcium) formed in the mill environment. Destabilization could liberate metals coprecipitated such as radium, uranium, and lead.

If a deep-lake system is chosen, natural processes within the lake can contribute to conditions such as meromixis. Such conditions mean that a further natural barrier isolating the tailings from the biosphere will develop and stabilize over time.

Future modification of the system will in all likelihood be limited to an alteration of the overlying sediments or water column. This will principally restrict only the upward movement of contaminants. Modification of downward migration would prove extremely difficult if not impossible.

There are a series of areas where further information and research is needed in order to address more completely any disposal option for uranium mine tailings. The ability to predict the behaviour of contaminants released from uranium mine tailings will be enhanced by further research in the following areas:

1. development of synthetic variables combining several processes to give resultant balances (e.g., relating biogeochemical mechanisms to physical transport);
2. a generic study of hypothetical tailings placements in a variety of lake systems;
3. chemical speciation and solid state structure of toxic elements at various pH and Eh levels;
4. studies of relations between anaerobic decay and dissolution of sulphide and other accessory minerals;
5. analysis of adsorption characteristics of contaminants onto different surfaces; (clays, organics, fracture linings, rock matricies, etc.);
6. pure solution studies on the chemistry of coprecipitation of toxic elements with minerals formed in diagenetic processes (e.g., vivianite);
7. studies on drainage basin erosion in relation to lake sedimentation;
8. further investigation of the causes and controls of natural meromixis;
9. verification and development of mathematical models for predicting meromictic stability;
10. development of unifying concepts for the prediction of bioconcentration, especially concepts relevant to the chemistry of soft-water Canadian Shield lakes;
11. coring of lake sediments that have received tailings runoff in the past to study distribution coefficients and to verify synthetic models and theories;
12. lake enclosure (limnocorral) studies of contaminant behaviour under various Eh and pH regimes;
13. studies of groundwater systems near existing ore bodies to better understand those factors affecting migration of radionuclides;
14. predictions of future atmospheric chemistry and response of tailings disposal systems to various environmental scenarios; and,
15. studies on defining limits beyond which it is unreasonable to expect that an assessment of particular variables (e.g., the effect humans may have on the weather) can be made with assurity.

Atomic Energy Control Board
P.O. Box 1046
Ottawa, Canada
K1P 5S9

Commission de contrôle de l'énergie atomique
C.P. 1046
Ottawa, Canada
K1P 5S9

INFO-0070(F)

ÉVALUATION QUALITATIVE DES PROCESSUS À LONG TERME RÉGISSANT LE COMPORTEMENT DES RÉSIDUS D'URANIUM DÉPOSÉS EN LACS PROFONDS

Étude effectuée pour la

COMMISSION DE CONTROLE DE L'ÉNERGIE ATOMIQUE

par

BEAK CONSULTANTS LIMITED
TORONTO, ONTARIO

mars 1982

RAPPORT DE RECHERCHE

RÉSUMÉ

Une Méthode acceptable d'évacuation des résidus provenant des usines de concentration d'uranium pourrait consister à les déposer dans des milieux de sédimentation caractéristiques des lacs profonds, en certains endroits au Canada. Comparé aux méthodes usuelles de gestion des résidus en milieu terrestre, ce moyen laisse entrevoir une meilleure stabilité à long terme des résidus. Du point de vue réglementaire, il s'avère cependant nécessaire de connaître les facteurs qui devraient être considérés pour évaluer l'acceptation d'une telle approache. Le rapport considère donc les variables d'ordre environnemental déterminant, à courtre et à longue échéance, le comportement des radionucléides et des éléments en trace dans les eaux souterraines, l'eau des lacs, et enfin dans la biosphère. Tous les facteurs physiques, chimiques et biologiques sont considérés. Les conclusions du rapport portent sur les avantages et les inconvénients de cette méthode d'évacuation.

Les opinions exprimées dans ce rapport sont celles des auteurs et n'engagent pas la Commission de contrôle de l'énergie atomique.

beak

AVANT-PROPOS

Le présent document examine et résume les points les plus importants d'un rapport présenté en annexe et portant le même titre, soumis par BEAK CONSULTANTS LIMITED à la Commission de contrôle de l'énergie atomique du Canada, en décembre 1981. L'annexe est à consulter pour les références bibliographiques, les tableaux et les explications détaillées. Le présent document renvoie d'ailleurs aux paragraphes utiles de l'annexe.

INTRODUCTION

Un des principaux problèmes environnementaux à résoudre pour pouvoir développer l'énergie nucléaire est la gestion à long terme des résidus de minerai d'uranium. Cette gestion doit être aussi sûre que possible car les résidus d'uranium contiennement des substances potentiellement toxiques, non seulement des traces d'uranium et de ses produits de désintégration, mais aussi d'autres métaux radioactifs ou chimiquement toxiques associés à l'uranium dans le minerai. Citons, entre autres, des radio-isotopes de la famille de l'uranium, comme U238, Th230, et Ra226, ainsi que du thorium, comme Th232, ainsi que des métaux lourds, As, Pb, Fe, Mn et Ni. La méthode couramment employée pour contenir les résidus, qui consiste à barrer une vallée, présente plusieurs limites fondamentales. L'érosion du terril peut entraîner le transport éolien de particules et le ruissellement de surface peut contenir des matériaux chimiques et radioactifs toxiques. La percolation des eaux de surface à travers la masse du terril peut lixivier les contaminants et les acheminer vers les eaux souterraines; dans certaines circonstances, ces contaminants pourraient même entrer en contact avec la biosphère. L'action des bactéries sur les pyrites en milieu non saturé entraîne une acidification qui accroît la dissolution des contaminants. Les végétaux qui poussent sur les terrils peuvent absorber par assimilation radiculaire, les éléments qu'il contiennent, qui peuvent ensuite pénétrer dans la chaîne alimentaire lorsque les animaux broutent.

À cause de ces problèmes, on envisage actuellement dans plusieurs endroits au Canada une autre possibilité, soit le dépôt dans des lacs profonds. En effet, des résidus placés au fond d'un lac se situent dans un environnement caractérisé par la sédimentation et non par l'érosion, à cause de la sédimentation continuelle de matières organiques et inorganiques produite dans le bassin de drainage du lac et de la protection contre l'érosion offerte par les eaux recouvrant les résidus. Par suite de la sédimentation continuelle, la couche de sédiments située entre l'eau et les résidus s'épaissit avec le temps. Si on suppose que le lac est suffisamment profond pour que les résidus soient bien au-dessous de la profondeur de compensation photosynthétique, c'est-à-dire au-dessous de laquelle les plantes ne croissent plus par manque de lumière, l'assimilation radiculaire par les macrophytes aquatiques ne présente plus de problème. Le dépôt en lacs profonds offre aussi l'avantage de protéger les résidus contre des perturbations météorologiques extrêmes comme les ouragans, les inondations et les sécheresses, par une couche d'eau permanente. Les bassins des lacs profonds peuvent être relativement isolés des réseaux d'eaux souterraines superficiels de sorte que les échanges se font surtout au niveau des couches d'eau supérieures du lac. Finalement, les sédiments ont tendance à être anoxiques, ce qui assure de bonnes conditions de stabilité géochimique pour beaucoup d'éléments à l'état de trace, qui se trouvent dans les résidus. Pour

ces raisons, les sédiments des lacs profonds sont généralement considérés comme des récepteurs plutôt que des sources pour les matières transportées dans un bassin versant. En d'autres mots, ce sont des zones d'où très peu d'éléments sont mobilisés et transportés dans les eaux en aval par rapport à la quantité totale des éléments présents.

Par contre, on doit envisager un certain nombre d'objections. Comme la plupart des lacs présentent une circulation totale de leurs eaux au moins une fois par an, tout contaminant qui peut migrer des résidus vers les eaux juste au-dessus pourrait être entraîné dans la zone trophogène et de là, dans les chaînes alimentaires aquatiques et terrestres. L'importance relative de ce mécanisme dépend du taux de lixiviation, de la direction et de l'amplitude du transport physique et des effets à long terme des perturbations. Il peut y avoir une deuxième objection, s'il existe une voie de migration vers les réseaux d'eaux souterraines profonds. Selon les caractéristiques spécifiques de chaque site, ceci pourrait être défavorable ou bénéfique (e.g. le transport vers les eaux superficielles plutôt que l'atténuation des contaminants dans la roche-mère en profondeur).

Avant de prendre une décision à l'égard d'un mode d'évacuation, il convient d'examiner qualitativement et, si possible, quantitativement tous les mécanismes et les voies de migration probables et de comparer les résultats d'évacuation à ceux d'autres modes d'évacuation. Ce document et le rapport qui l'accompagne constituent un guide décrivant un cadre conceptuel et considérant certaines variables spécifiques dans le cadre des propositions de dépôt en lacs profonds.

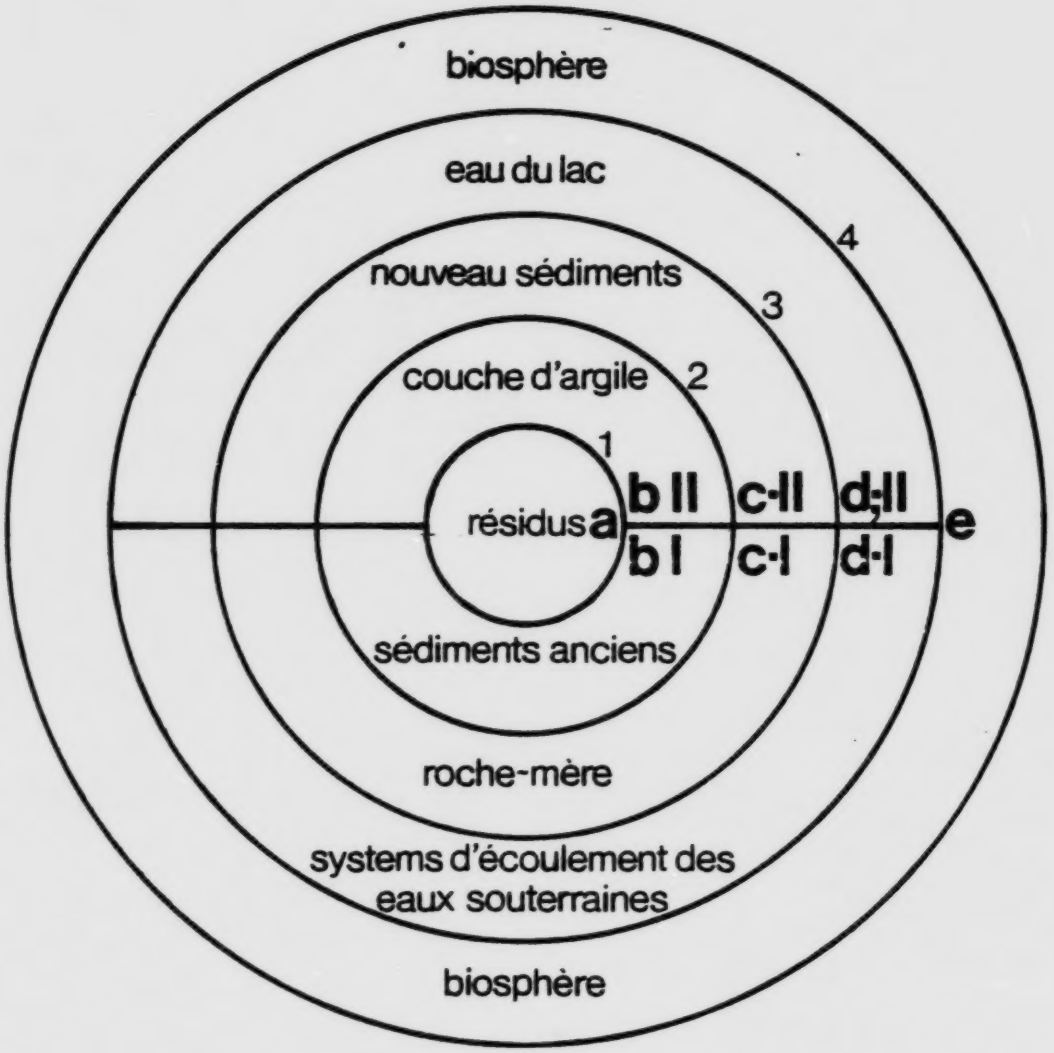
APPROCHE

Nous avons choisi d'aborder le problème dans l'ordre croissant c'est-à- dire en considérant d'abord les caractéristiques des matières contenues dans la masse de résidus pour en arriver à la biosphère en général, après avoir franchi diverses barrières. Une série de cercles concentriques (Figure 1) illustre bien notre propos. La masse des résidus figure au centre. Chaque bande (A à E) représente l'emplacement d'un contaminant en migration à un stade donné de sa progression qui résulte de divers processus physiques, chimiques ou biologiques. Chaque cercle (1 à 4) représente une barrière ou une interface entre un emplacement et le suivant. Une ligne horizontale divise le diagramme en deux moitiés représentant les deux directions possibles de migration: vers le bas, dans le système roche-mère et eaux souterraines; et vers le haut, dans l'eau sus-jacente. Les deux moitiés se réunissent dans la bande la plus externe qui représente la biosphère dans son ensemble.

À chaque bande ou interface, une évaluation des variables impliquées donnera une prévision du mouvement des contaminants qui pourra s'exercer vers l'extérieur de la masse des résidus, vers l'intérieur, ou être négligeable. Deux échelles de temps ont été considérées. Pour la période de 0 à 100 ans après le dépôt des résidus, les processus à court terme (généralement irréversibles) sont les plus importants, car l'état géochimique final des résidus peut encore évoluer, dans la mesure où la consolidation physique n'est pas encore complétée et que les limites biochimiques sont toujours fluides. Pendant la période de 100 à 10 000 ans, les phénomènes tels que les changements cycliques du climat, les événements sismiques, les événements météorologiques rares, ainsi que la lente diffusion deviennent de plus en plus importants et l'on ne peut faire que les prévisions les plus générales. La période au-delà de 10 000 ans est considérée comme impondérable.

Figure 1

Schéma conceptuel des principales barrières entre les résidus et la biosphère



Interfaces

1 eau interstitielle de la masse de résidus
2 eau interstitielle en bordure des résidus-l'eau interstitielle des sédiments
3 eau interstitielle des sédiments-eau du lac
4 eau du lac-chaîns alimentaires aquatiques

"A" - MASSE DES RÉSIDUS (Sous-section 1.1 de l'annexe)

La masse des résidus est la partie la plus importante du schéma de la Figure 1. Ce n'est que si un contaminant s'échappe de la zone A que l'on se préoccupera des interfaces et bandes suivantes. En outre, les contaminants dans la masse de résidus sont dans un espace relativement confiné donc plus facilement sujet à une modélisation visant le contrôle et la prévision de leur comportement. De la même manière, on peut considérer que chaque bande ou cercle est plus critique que le suivant.

Une fois placée au fond du lac, las masse des résidus sera formée de matières particulaires et d'eau interstitielle. La stabilité physique de l'environnement profond du lac nous permet de postuler que la matière particulaire ne sera pas mobile aussi longtemps que la masse finale de résidus repose complètement dans la zone de dépôt du lac, à l'abri de toute force d'érosion importante. La première étape pour prédire la migration des éléments toxiques est de connaître la structure et le contenue de la masse des résidus. Par exemple, un type de résidus peut comprendre:

a) la gangue lixiviée (minerai broyé et lixivié);

b) les matériaux sorbés ou précipités sur les minéraux des résidus; et

c) les boues ou précipités chimiques.

Ces trois composants des résidus ont différentes concentrations de contaminants et différentes caractéristiques de lixiviation.

La deuxième étape sera de connaître la capacité de lixiviation des contaminants (passage de la forme particulaire à l'eau interstitielle). La lixiviation sera fonction de la surface des particules exposée à l'eau interstitielle et des processus géochimiques et biologiques contrôlant le partage des contaminants entre les formes solubles et particulaires. Puisque les propriétés géochimiques et biologiques-chimiques peuvent être différentes d'un élément à l'autre, on doit considérer individuellement chaque contaminant potentiel.

Les variables géochimiques influençant le partage des éléments sont:

a) le pH, responsable de la formation d'ions et de coefficients de dissolution;

b) l'Eh (potentiel d'oxydo-réduction ou potentiel redox) qui détermine si les ions seront sous forme oxydée ou réduite;

c) la force ionique qui influence l'activité chimique des ions dissous et donc leur capacité à former des précipités;

d) le système tampon qui contrôle le pH; et

e) les formes cristallines des minéraux qui peuvent se dissoudre pour

changer les concentrations de l'eau interstitielle et peuvent précipiter s'il y a sursaturation.

Les variables biochimiques importantes sont:

a) l'existence de sources de carbone et d'oligo-éléments, nécessaires pour les communautés bactériennes;

b) les communautés microbiennes influencent le potentiel redox et le pH; et

c) les agents chimiques de lixiviation peuvent provoquer la désorption des métaux et peuvent être produits par les bactéries et les végétaux supérieurs.

À long terme, le pH, le Eh et la force ionique dans la masse de résidus sont fonction:

a) des taux relatifs de dissolution de divers minéraux comme un arséniure dans la gangue, et des précipités chimiques formés dans l'usine comme le gypse, le jarosite, la calcite;

b) des espèces chimiques entrant dans la masse de résidus avec l'eau souterraine; et

c) du fait que des communautés microbiennes particulières se développent ou non dans la masse de résidus.

Par exemple, le pH peut diminuer à cause du développement d'une communauté oxydant le soufre, quand toute la calcite a été dissoute et transportée à l'extérieur de la masse de résidus. A un faible pH, des quantités croissantes de gypse dissolvent et augmentent la force ionique. Donc, bien que des variables comme le pH et le Eh soient des variables principales, elles sont déterminées et contrôlées par le système géochimique et biochimique particulier qui domine à un moment donné. La température est considérée comme une variable principale à la fois pour les processus géochimiques et pour les processus biologiques, car elle commande la vitesse des réactions organiques et inorganiques.

Selon les combinaisons de valeurs de ces variables et d'autres on peut faire un certain nombre de scénarios bien différents. Par exemple, dans une masse de résidus ne contenant pas de matière organique, l'ion sulfure (S^{--}) et un ion métallique (M^{++}), contaminant potentiel, peuvent passer dans l'eau interstitielle et s'échapper de la masse de résidus. Toutefois, s'il y a suffisamment de fer réduit (Fe^{++}) provenant de réactions organiques, le fer peut coprécipiter avec le sulfure et le contaminant potentiel, et réduire ainsi le taux de pertes des résidus. Dans ce dernier cas, l'effet confiné des variables géochimiques (fer présent) et biochimiques (matière organique, bactéries) modifie le partage.

Une fois que l'on a défini l'état géochimique des résidus, on peut estimer des coefficients de partage entre les particules et l'eau interstitielle. L'étape suivante consiste à examiner le transport dans l'eau interstitielle.

TRANSPORT DANS L'EAU INTERSTITIELLE

La deuxième zone de la Figure 1, par ordre d'importance, est le cercle 1 qui représente les processus de transport dans l'eau interstitielle. Ce sont les processus par lesquels les contaminants en solution peuvent atteindre la zone B de la Figure 1 (vers le haut ou vers le bas selon les gradients de transport). À court terme, les résidus peuvent être considérés comme une "boîte noire" traversée par de l'eau. Une fois que le coefficient de partage entre les particules et l'eau interstitielle est établi, et si la concentration de l'eau est constante à un point donné de la masse de résidus, le transport est régi par deux mécanismes physiques:

a) le transport advectif, mouvement de l'eau dans une direction (fonction du gradient hydraulique dans la masse de résidus et de la perméabilité du milieu); et,

b) les processus de type diffusion, limités par les gradients de concentration et provoqués par le processus de physique moléculaire de collision aléatoire avec les molécules d'eau, ou la dispersion qui est provoquée par la diffusion moléculaire, l'advection et la turbulence.

Les variables à évaluer comprennent les coefficients de perméabilité directionnelle, les gradients hydrauliques, les gradients de concentration entre l'eau interstitielle et celle des strates sous-jacentes et sus-jacentes et le coefficient global de diffusibilité.

Les processus à long terme touchant le transport physique comprennent la consolidation de la masse de résidus (modifica tion de la perméabilité et de la diffusibilité globale), les changements dans les gradients hydrauliques causés par les processus géologiques à long terme et peut-être par les perturbations anthropiques, les changements dans les gradients chimiques et la réponse de la masse de résidus à l'accélération soudaine causée par des mouvements sismiques ou des événements météorologiques.

"B-I" - SÉDIMENTS LACUSTRES EXISTANTS (Sous-section 1.3 de l'annexe)

Les principes décrivant le rôle de sédiments lacustres sur les contaminants des résidus sont discutés plus loin (Sous-section 3.2, Nouveaux sédiments). Voici les différences importante entre les sédiments lacustres existants (dépôts antérieurs), situés au-dessous de la masse de résidus, et les nouveaux sédiments qui couvrent les résidus:

a) si les variables générales de vieillissement du bassin versant restent constantes, les sédiments existants seront consolidés et leurs coefficients de perméabilité et de diffusibilité globale seront réduits en comparaison des sédiments postérieurs; leurs valeurs seront encore

diminuées par le poids additionnel des résidus déposés au-dessus;

b) les sédiments existants ont tendance à contenir des matières organiques plus anciennes et plus réfractaires, ce qui fait que leur potentiel pour supporter une communauté microbienne est inférieur; et

c) les sédiments existants seront physiquement isolés des eaux du lac et, de ce fait, ne seront pas l'objet de perturbation physique (bioturbation) par des organismes aquatiques.

"B-II" - BARRIÈRE POUR LES RÉSIDUS (Sous-section 1.2 de l'annexe)

Une barrière installée au-dessus des sédiments lacustres existants ou des résidus déposés, représente un écran physique et chimique empêchant la mobilisation des contaminants de la masse de résidus et le mouvement de ces contaminants à l'extérieur de la masse. La capacité de cette barrière à retarder le transport, à la fois en ralentissant les processus géochimiques et en restreignant l'écoulement advectif, est donc de première importance. Comme dans la masse de résidus, il y a des processus de dissolution et d'adsorption chimique, et des phénomènes de transport physique par advection et diffusion. Parmi les barrières possibles, citons les revêtements composés exclusivement ou partiellement d'argiles.

La composition chimique de la barrière déterminera le type d'ions H^+ ou S^{--}, par exemple, et les surfaces que rencontrera un contaminant soluble. Les particules d'argile tendent à avoir de bonnes propriétés d'adsorption de certaines radionucléides et métaux lourds, tandis que l'apatite collophane à grain fin est un excellent adsorbeur de radium.

La composition chimique de l'eau interstitielle de la barrière est aussi importante. En présence de conditions anoxiques, la précipitation des sulfures métalliques peut créer une barrière supplémentaire à la migration. La présence d'éléments capables de former d'autres précipités avec les contaminants anticipés pourrait être délibérément recherchée dans la barrière pour améliorer encore la rétention de ces contaminants.

Comme dans la masse des résidus elle-même, le transport de l'eau interstitielle dans la barrière est commandé par la perméabilité, la diffusibilité, ainsi que l'abondance et la direction des gradients hydrauliques et chimiques. La diffusion moléculaire dépend des gradients de concentration ionique entre l'eau interstitielle de la barrière et l'eau interstitielle des sédiments. La perméabilité et la diffusivité globale seront influencées par l'importance de la consolidation de la barrière. La consolidation est un processus de longue durée qui provient du tassement des particules dans les espaces vides. Elle est causée par la masse compacte de résidus au-dessus et, dans le cas d'une barrière sous-jacente, par toute la masse de résidus elle-même. La compaction de l'argile est très influencée par les charges de surface sur les particules d'argile. Une barrière composée de particules de charge uniforme ne se tasse pas aussi bien qu'une barrière constituée de réseaux alternés de charges positive et négative. L'épaisseur de la couche sera une variable principale

déterminant le temps de passage du contaminant pour un ensemble donné de conditions physiques et chimiques.

La stabilité de la barrière doit aussi être considérée. Il faut évaluer la cohésion de la couche qui est une variable à court terme. Avant que de nouveaux sédiments se soient accumulés, la couche peut être exposée à des forces de cisaillement causées par les courants de fond, spécialement si des extrêmes météorologiques se produisent pendant cette phase, ou par des forces gravitationnelles si la barrière n'est horizontale. Un tassement et une reconsolidation rapides de la masse de résidus après de telles perturbations physiques peuvent entraîner les augmentations relativement mineures du transport des contaminants par rapport aux processus de transport constant et de longue durée. A long terme, il faut considérer aussi la résistance aux perturbations provoquées par des phénomènes sismiques.

"C-I" - ROCHE-MÈRE (Section 3 de l'annexe)

Si les contaminants dissous quittent la masse de résidus et migrent à travers les sédiments lacustres et les débris glaciaires sous-jacents, ils peuvent atteindre la roche-mère (zone C-I, dans la figure 1). Le comportement des contaminants dans la roche-mère est fonction principalement de sa structure physique, de sa nature physique et de sa stabilité. La structure physique déterminera l'importance relative de l'écoulement en milieu poreux, de l'écoulement le long des fractures et de la diffusion au transport total des contaminants. Les variables importantes sont le type de roche-mère, la perméabilité de la matrice, la diffusibilité ainsi que la présence, la taille, la densité et la continuité des fractures. Comme avec les barrières, la nature chimique du revêtement des fractures de la roche-mère et de la matrice de la roche déterminera sa capacité d'adsorption. Si les coefficients globaux de perméabilité de diffusibilité sont faibles et si la capacité d'adsorption est suffisamment élevée, on peut considérer la roche-mère comme un récepteur géologique à long terme pour de nombreux radionucléides et autres contaminants.

À long terme, la stabilité physique et chimique de la roche-mère est un élément critique pour la sécurité du système. L'instabilité conduirait principalement à un transport plus rapide dans les réseaux d'eaux souterraines, bien que l'effrondement physique du système de retenue et peut-être de la masse de résidus elle-même ne puisse être écarté dans certaines formations.

Les variables permettant d'évaluer la stabilité physique sont:

a) le type de roche-mère sédimentaire, métamorphique ou ignée;

b) les antécédents d'activité sismique;

c) la présence et la nature des fractures - guide de l'histoire géologique récente et de la stabilité;

d) la stabilité à long terme des fractures - réaction aux futurs changements des contraintes; et

e) le taux de rebond isostatique post-glaciaire et de mouvement des plaques tectoniques - leurs relations avec le système de fractures de la roche-mère, la distribution des fractures verticales et l'ouverture des fractures.

La réactivité chimique peut être évaluée en étudiant:

a) la nature du revêtement des surfaces des failles et son potentiel pour retarder géochimiquement le taux effectif de migration des contaminants;

b) la minéralogie de la matrice de la roche-mère et son potentiel pour retarder par adsorption la migration des contaminants qui se diffusent dans cette matrice;

c) le potentiel d'interaction entre les matériaux migrants de la masse de résidus et la surface des fractures. Par exemple, un lixiviat acide des résidus, migrant dans des fractures revêtues de carbonate, entraînera la dissolution des carbonates et l'agrandissement des fractures, augmentant ainsi le taux de transport des contaminants.

"C-II" - NOUVEAUX SÉDIMENTS (Sous-sections 1.3, 2.2 et 4. de l'annexe)

Qu'une barrière artificielle existe ou non, la sédimentation naturelle de la matière particulaire du lac et du bassin, crée une barrière supplémentaire entre la masse des résidus et l'eau du lac. Bien que la translocation physique et biologique de la matière particulaire soient une possibilité, spécialement dans les sédiments en eau peu profonde, il est néanmoins raisonnable de supposer qu'à long terme, en lac profond, la plupart des mouvements des contaminants se font sous forme soluble. Pour ces constituants solubles, l'équilibre chimique et le transport physique jouent des rôles clés.

Comme dans la masse des résidus, l'advection verticale et la diffusion sont les principaux mécanismes de transport. L'advection sera influencée par les pressions hydrauliques (écoulement de l'eau souterraine et expulsion de l'eau interstitielle due à la consolidation). La minéralogie, la granulométrie, la porosité, la masse volumique et la teneur en matières organiques sont des variables utiles pour estimer le potentiel de transport physique et chimique. La densité et la composition de la population d'invertébrés benthiques déterminent les mouvements de l'eau d'origine biologique dans les sédiments supérieurs. Les variables reliées à la diffusion comprennent les gradients chimiques, l'adsorption et la porosité; ces variables peuvent être formulées en un coefficient de diffusion effectif pour les sédiments. Les gradients chimiques d'autres éléments peuvent être importants pour le mouvement des contaminants s'ils influencent la force ionique et entrent en compétition pour les sites d'adsorption. Les caractéristiques de consolidation des nouveaux sédiments s'accumulant au-dessus des résidus influencent à la fois l'advection et la diffusion, ainsi que la stabilité à long terme. Les variables clés dans la détermination des caractéristiques de consolidation sont la résistance à l'effort de cisaillement des sédiments, la densité des macro-invertébrés, et la caractère floculant des particules.

Les processus biogéochimiques à l'intérieur des nouveaux sédiments détermineront les concentrations en éléments solubles disponibles pour le transport physique à travers cette barrière de sédiments en croissance. Les processus géochimiques critiques comprennent:

a) le dépôt de métaux lourds, principalement comme composés inorganiques inertes;

b) la précipitation sous forme d'hydroxydes, de carbonates, de sulfures et de phosphates, en particulier dans les eaux alcalines;

c) la sorption et l'échange cationique à la surface des particules (argiles, oxydes de Fe et de Mn, matières humiques, etc.);

d) la complexation ou la chélation des métaux dissous par les composés organiques dissous (acides humiques et fulviques);

e) la coprécipitation des métaux avec les hydroxydes, les carbonates et les sulfures.

Les taux relatifs et l'ampleur de ces processus doivent être déterminés avant de faire des prédictions. Les variables essentielles pour ces calculs sont:

a) les propriétés des sédiments:

1) la granulométrie des particules - gouverne la perméabilité, la porosité et la surface disponible pour les réactions avec les solutés;

2) les conditions chimiques de la surface des particules - déterminent les isothermes d'adsorption et la capacité d'échange de cations et donc la capacité d'atténuation des sédiments.

b) les conditions chimiques de l'eau interstitielle:

1) le pH - influe sur les taux de dégradation des minéraux, d'adsorption, de formation de complexes, de solubilité et de précipitation;

2) le pouvoir tampon - contrôle le pH, la précipitation;

3) l'Eh (potentiel redox) - contrôle la vitesse et la direction des réactions, et les espèces métalliques formées;

4) les concentrations des principaux ions - influencent les réactions biochimiques et les précipités possibles;

5) les concentrations des contaminants - un moyen de surveillance pour le contrôle des gradients de diffusion.

Les processus biochimiques déterminent aussi les coefficients de distribution

des contaminants dans l'eau interstitielle. En plus d'influencer la capacité d'échange de cations des sédiments, l'abondance de matières organiques détermine aussi quelle population microbienne pourra proliférer dans les sédiments. Ces systèmes biologiques, à leur tour, déterminent la valeur du potentiel redox, une des variables chimiques les plus importantes. La demande d'oxygène des sédiments peut diminuer les concentrations d'oxygène au point où le potentiel redox tombe, entraînant la remise en solution de certains oxydes métalliques et des métaux coprécipités. A l'opposé de cette tendance, se trouve la capacité des sulfures de former des précipités métalliques à des potentiels redox négatifs. La plupart des sédiments devraient avoir un potentiel redox négatif ou neutre.

Un autre série de mécanismes, mal compris actuellement, implique la capacité de divers composés organiques (matières humiques, amioacides, etc.) à former des complexes qui peuvent entrer en concurrence avec des particules inorganiques pour les métaux dissous et donc augmenter ou diminuer leur solubilité. L'absorption directe des métaux par les microbes et la bioturbation provoquée par les macro-invertébrés ajoutent à la complexité du système réuni des sédiments, de l'eau interstitielle et du biote. Finalement, la stratification des populations biologiques, les variables chimiques et la structure physique amène les sédiments à former une barrière multicouches qu'il est difficile de considérer comme un tout homogène.

En dépit de la complexité physique, chimique et biologique de ces mécanismes, une variables prédomine, soit l'épaisseur de la couche de sédiments. Comme les sédiments s'accumulent avec le temps au-dessus de la masse des résidus, c'est la vitesse de sédimentation qui est le facteur essentiel. Les caractéristiques d'érosion dans le bassin de drainage du lac (e.g. apports de matières organiques et inorganiques allochtones) et la productivité du lac (e.g. matières organiques autochtones) sont d'un intérêt primordial pour prédire les vitesses de sédimentation.

"D-I" - EAUX SOUTERRAINES PROFONDES (Sections 3 et 4 de l'annexe)

Le réseau d'eaux souterraines est une composante critique car, étant l'avant-dernière bande (zone "D-I") de la figure 1, les contaminants qui en sortent ont la possibilité de se répandre directement dans la biosphère. Donc, les principales questions à se poser sont:

a) de quelle manière le réseau d'eaux souterraines sous la masse des résidus est-il relié aux eaux de surface,

b) quels sont les facteurs de ce réseau aquifère qui agissent sur:

1) le mouvement des eaux,

2) le degré d'atténuation des contaminants, et

3) le degré de dispersion et de dilution des contaminants.

Pour répondre à ces questions, il faut connaître les modèles régionaux de

réseaux d'eaux souterraines et estimer la perméabilité des fractures et de la matrice, les pressions hydrauliques, les coefficients de diffusion et la capacité d'adsorption du milieu. Après avoir tenu compte du temps de transport et de l'adsorption pendant le transport, on peut estimer un taux final de décharge de contaminants, ou "taux de pertes", et le comparer au niveau acceptable de libération de contaminants. La stabilité du réseau d'eaux souterraines est une variable principale à long terme, et elle est fonction de la stabilité de la roche-mère (voir "C-I"), des cycles climatiques à long terme influençant le niveau de la nappe phréatique (section 6 de l'annexe) et des perturbations induites par les activités humaines (section 7 de l'annexe).

"D-II" - EAUX DU LAC (Sous-sections 2, 6.3 et 6.4 de l'annexe)

La masse d'eau du lac est à deux cercles du centre de la figure 1. C'est la dernière ligne de défense avant une entrée éventuelle dans le milieu environnant et c'est donc le milieu qui réagit physiquement et biologiquement avec la biosphère. Par conséquent, même si on considère ces eaux comme une ligne de défense, les concentrations de contaminants qui s'y trouvent préoccupent directement les gestionnaires en raison des activités humaines qui permettent un contact avec ces eaux.

Pour une vitesse donnée de transport d'un contaminant de la masse de résidus à l'eau du lac, un certain nombre de mécanismes peuvent intervenir dans la disponibilité biologique. A l'interface des sédiments et des eaux, le potentiel redox peut déterminer si un élément passera dans l'eau. De forts courants de fond peuvent augmenter les pertes de contaminants en agitant les sédiments et en augmentant les gradients chimiques dans l'interface. Le système de circulation vertical dans le lac est d'un intérêt plus grand. Le transport advectif est fonction de cette circulation verticale de l'eau. La plupart des lacs des latitudes tempérées sont stratifiés par des gradients de densité induits thermiquement pour la plupart de l'année, mais subissent un mélange vertical complet (circulation) au moins une fois par année (lacs monomictiques) et habituellement deux fois (lacs dimictiques). Typiquement, la circulation d'automne se produit quand la stratification estivale on orthograde (eaux plus chaudes et plus légères près de la surface et eaux plus froides et plus denses en dessous de 4°C près du fond) se rompt. Ceci se produit quand la température plus fraîche de l'air amène les eaux de surface à se refroidir et leur température à se rapprocher de celle des eaux du fond. La stratification hivernale est rétrograde (eaux plus légères près de 0°C à la surface, eaux plus denses près de 4°C au fond) et se rompt lorsque les températures plus chaudes du printemps réchauffent les eaux de surface.

Comme nous l'avons dit (voir Introduction), les périodes de circulation sont les moments où se produit le transport vertical advectif des métaux échappés des sédiments qui passent des eaux de fond aux eaux de surface. Les caractéristiques de circulation du lac que l'on peut mesurer sont:

a) la fréquence et la durée des circulations - nombre de jours par année où les eaux de fond et de surface se mélangent;

b) la profondeur maximum de la circulation verticale - le mélange est complet jusqu'au fond du bassin le plus profond;

c) le taux d'érosion de la thermocline (gradient de densité) pendant la stratification estivale - susceptibilité au brassage par les vents, mesuré par les profils verticaux de température;

d) la stabilité de la stratification (par exemple, nombre de Richardson) ou la résistance à la force de cisaillement induite par le vent.

En plus des gradients de densité thermique, les gradients de salinité contribuent aussi à la stabilité verticale dans certains lacs (voir sous-section 7.2 de l'annexe). Si un lac est suffisamment profond et que ses eaux de fond sont plus salées que ses eaux de surface, la force de cisaillement induite par le vent peut ne pas être capable de modifier les différences de densité verticale même en présence d'isothermes verticaux. Ces lacs sont appelés méromictiques. Les sources de salinité comprennent les incursions d'eaux de mer dans les lacs côtiers pendant les orages (extogénie), la dissolution des dépôts géologiques (crénogénie) et la décomposition de matières organiques (biogénie). Puisque la méromicticité empêche le transport advectif vers les eaux de surface et comme l'eau saline réduit le coefficient de diffusibilité, cette condition est souhaitable dans un système de dépôt de résidus. Dans certains cas, notamment au lac Garrow (T. N.-O.), des lacs méromictiques naturels peuvent être proches d'un site minier, faisant d'eux des sites de dépôts potentiels. Actuellement, seul un nombre limité de lacs méromictiques ont été étudiés au Canada, mais il y en a sans aucun doute beaucoup d'autres non découverts. Lorsque l'on cherche des lacs méromictiques, les variables à mesurer comprennent:

a) le gradient vertical de salinité (total des matières dissoutes): (picnocline);

b) la stratification thermique durant toute l'année;

c) le réchauffement adiabatique des eaux profondes;

d) la prolifération des bactéries réductrices du soufre en eaux profondes; et

e) le rapport entre l'exposition au vent et la distance verticale d'eau à être mélangée.

Une autre option consiste à essayer d'induire artificiellement la méromicticité en ajoutant des sels dans les eaux profondes. Le calcul du gradient de salinité requis pour produire un degré donné de stabilité verticale est relativement simple, mais la méromicticité est produite seulement par la combinaison convenable de salinité, de profondeur relative du lac et de protection par des collines contre la force de cisaillement induite par les vents. Le brassage causé par le vent est difficile à prédire à cause de notre manque de connaissances sur les effets du vent sur la couche en contact avec l'atmosphère

et à cause de l'incertitude quant à la fréquence des tempêtes accompagnées de vents violents. Les facteurs spécifiques comprennent la présence de collines environnantes, la longueur et la largeur du lac, et la direction des vents dominants. Si l'on choisit de créer une chimiocline artificielle dans le lac, il faut prendre soin de s'assurer de la stabilité de la méromicticité provoquée. L'eutrophisation du lac et, donc, la création possible d'une méromicticité stabilisée biogéniquement à long terme est une de ces possibilités.

La diffusion verticale est aussi un facteur critique pour le transport vers le haut. Des modèles mathématiques permettant de prédire la diffusion en fonction des gradients de température sont en cours d'élaboration. Les coefficients de diffusion turbulente tendent à être supérieurs lorsque la stabilité thermique est faible.

Les facteurs biologiques et chimiques contrôlant les coefficients de distribution entre les particules et les solutés dans la colonne d'eau sont d'une grande importance pour prédire la remontée nette des contaminants dissous échappés de la masse des sédiments. Les ions métalliques qui deviennent particulaires tendent à se déposer à nouveau dans les sédiments tandis que les ions solubles sont sujets à un transport physique et à une absorption biologique potentielle. Les mécanismes importants de minéralisation et de précipitation sont:

a) la précipitation ionique - lorsque la concentration d'un métal en solution dépasse la limite de solubilité, ou lorsque des ions métalliques sont absorbés sur des particules inorganiques;

b) l'absorption biogénique - consommation des métaux par les organismes, principalement par le phytoplanction, qui se déposent ensuite sur les sédiments; et,

c) l'adsorption sur les composés humiques - fixation des ions par de grosses molécules organiques. Ceci est spécialement important dans les lacks dystrophiques du Bouclier canadien.

Les variables chimiques utiles pour prédire les coefficients d'adsorption et de distribution et la probabilité de la précipitation ionique sont:

a) le pH - règle la solubilité, dépend dans beaucoup de lacs du système tampon représenté par les bicarbonates. Dans d'autres lacs, il peut être fixé par d'autres minéraux ou matières organiques;

b) les régimes d'oxygène - déterminent les populations microbiennes à l'interface des sédiments et de l'eau; ces populations commandent les potentiels redox;

c) les carbones organiques particulaires (COP) et dissous (COD), substrats potentiels pour la croissance bactérienne et surfaces pour l'adsorption chimique et le dépôt subséquent; ils influencent la consommation d'oxygène;

d) les principaux anions et cations - matières nutritives potentielles pour la croissance biologique qui fournissent la preuve d'apports inorganiques allochtones;

e) les concentrations de fond de contaminants - serviront à indiquer l'importance de la containation des résidus par rapport aux conditions originales; et

f) les facteurs de productivité biologique (N, P, Si, métaux oligo-éléments, vitamines, composés organiques) - d'intérêt potentiel pour la précipitation biogénique et l'induction ou le maintien de la méromicticité.

Les variables biologiques nécessaires pour prédire le transport ascendant net comprennent:

a) la structure de la communauté bactérienne (aérobie ou anaérobie);

b) le phytoplancton,capacité d'adsorption et de consommation de métaux;

c) le zooplancton, capacité de concentration et de dépôt des métaux pour permettre leur entrée dans les chaînes alimentaires et pour régénérer les matières nutritives pour la croissance des phytoplanctons;

d) le benthos - bioturbation des sédiments supérieurs; et

e) les poissons - capacité de recyclage des aliments nutritifs, de consommation du zooplancton et du benthos, et de fournir un maillon important dans la chaîne alimentaire de l'homme.

Dans une analyse finale de la mobilisation des contaminants, le taux de sortie des contaminants de la masse de résidus à travers les sédiments est considéré comme l'entrée initiale pour une prédiction mathématique, et les mécanismes de transport vertical sont évalués par rapport aux taux de minéralisation et de sédimentation de façon à prédire la perte de contaminants vers les eaux de surface. La circulation horizontale peut mener au transport vers les exutoires du lac ou les zones moins profondes.

"E" - BIOSPHÈRE (Sous-sections 1.3, 2.1, 5 et 7 de l'annexe)

La principale préoccupation dans tout problème d'élimination de contaminants est celle de la protection de l'homme et des organismes auxquels il attache de l'importance, pour des raisons alimentaires, esthétiques ou écologiques.

Les premiers organismes à incorporer les contaminants provenant de la masse de résidus en lacs profonds seraient le plancton microscopique. Ces organismes n'ont pas en eux-mêmes une très grande valeur, mais ils sont au début de la chaîne de nourriture qui culmine avec les organismes supérieurs qui entrent souvent dans la nourriture de l'homme. Le problème est accru par la tendance de

certains éléments à l'état de trace, dans certains systèmes biologiques, à devenir de plus en plus concentrés au fur et à mesure que les niveaux trophiques sont plus élevés. On estime cette concentration par le facteur de bioaccumulation (FB), qui correspond au rapport entre la concentration d'un contaminant dans les organismes et la concentration dans l'eau. Si l'on peut prédire les concentrations dans les eaux de surface, on peut utiliser les valeurs du FB pour estimer les concentrations dans la chair et la toxicité résultante chimique ou radiologique. Une difficulté rencontrée au Canada est que la plupart des FB disponibles dans la documentation ont été observés dans des lacs de systèmes calcaires, tandis que beaucoup de lacs au Canada sont dans le bouclier précambrien, où les conditions chimiques et la biote de l'eau, et donc le comportement des contaminants, sont tout à fait différents. L'application des FB d'eaux dures à ces lacs n'est probablement pas valable.

En plus de la bioaccumulation dans le lac lui-même, la perte de contaminants tels que les métaux par la décharge du lac vers les eaux en aval ou les réseaux d'eaux souterraines sont deux autres voies d'entrée dans la biosphère. Leur importance relative dépend du biote qui se trouve sur leur passage.

Les effets de la biosphère sur le lac doivent aussi être évalués (section 5 de l'annexe). En particulier, il faut considérer les apports de matériaux du bassin de drainage vers le lac. Dans la plupart des lacs, la principale source d'éléments nutritifs (pour la croissance des phytoplanctons et la précipitation biogénique des contaminants), de sels minéraux (capacité tampon, précipitations chimiques), et de matières humiques (complexation, chélation des métaux) est le bassin versant. De plus, un grand nombre de lacs canadiens appartiennent à la catégorie dystrophique, pour laquelle la plupart des matières organiques du lac sont d'origine allochtone. Dans ces lacs, la force essentielle pour les populations bactériennes est l'énergie fixée dans le bassin versant et transportée au lac par les cours d'eau, le ruissellement et les eaux souterraines. Finalement, les apports du bassin de drainage déterminent en grande partie les niveaux de fond de radionucléides qui serviront à comparer l'importance des contaminants en provenance de la masse de résidus.

Pour ces raisons, il est nécessaire d'estimer les apports de matériaux dans le lac. Ils peuvent être exprimés en unités de masse/temps, masse/surface/temps, ou calculés à partir des débits et des concentrations de fond. Les variables à court terme dans ces calculs comprennent la composition de la roche et du sol, la solubilité et la capacité de lixiviation, la productivité terrestre, l'utilisation des terres, le volume et les conditions chimiques des précipitations, la pente du bassin et des caractéristiques de l'érosion. Les prévisions des changements à long terme dans l'apport de matériaux dépendent de variables comme les futures utilisations des terres, l'activité sismique ainsi que les cycles et les extrêmes météorologiques et les changements hydrologiques (spécialement anthropiques; voir sous-sections 6.1 à 6.4 et 7.1 de l'annexe).

En dehors des unités hydrologiques bien définissables du lac et du bassin de drainage, il est très difficile d'évaluer l'importance relative des principales variables atmosphériques et des matières dont elles influencent le lac, son bassin et la biosphère. Le bassin de drainage du lac est influencé par des phénomènes atmosphériques (sous-section 6 de l'annexe) comme le rayonnement

solaire, les vents, les précipitations (transport des contaminants à grande distance et acidification anthropogène), les cycles de température et la composition gazeuse. Tous ces facteurs sont sujets à des changements à très long terme. Finalement, les effets que les humains peuvent avoir sur des lacs ou des bassins de drainage particuliers et sur la biosphère en général sont très difficiles à prédire. Les variables à court terme comprennent l'utilisation des terres, la pêche et la chasse récréatives, et les perceptions changeantes du public sur ce qui est acceptable pour l'environnement. Parmi les possibilités à long terme, on trouve les changements dans la répartition des habitations, dans l'atmosphère en raison des activités humaines (cycle du CO_2), les feux de forêts, l'exploitation des forêts, l'eutrophisation des lacs, l'acidification et les manipulations hydrologiques (barrages, réservoirs, etc.).

La question de savoir où les effets de la biosphère sur la masse de résidus (et vice versa) deviennent trop diffus et difficiles à prédire doit être résolue par l'organisme de réglementation responsable des décisions de gestion de résidus particuliers. Par exemple, dans un lac relativement petit, sujet à une utilisation récréative intense, les turbulences causées par les moteurs hors-bord doivent être considérées car elles peuvent contribuer au transport advectif vers le haut des contaminants provenant des résidus et à l'effort de cisaillement sur une barrière récemment placée sur les résidus. Pour un lac de taille et de forme semblables, situé dans une région plus inaccessible, ce type de perturbation serait probablement négligeable, mais des prévisions à long terme doivent prendre en considération les effets possibles des changements futurs dans la population et des progrès technologiques.

CONCLUSION

Cette revue des nombreux mécanismes et processus qui contribuent à la mobilisation et la distribution des contaminants en provenance des résidus d'une mine d'uranium, placés dans un lac profond, amène à une série d'énoncés et de conclusions:

1. Dans un contexte géologique à long terme et dans le but de minimiser le taux de dispersion des contaminants, il est préférable de placer les résidus de mine dans des dépressions locales ou régionales qui sont des milieux de sédimentation plutôt que sur des hauteurs topographiques qui sont des lieux d'érosion.

2. Dans les principales régions de mines d'uranium au Canada, les précipitations relativement élevées, la mince couche superficielle, la faible perméabilité de la roche-mère et la topographie, font qu'il existe des dépressions naturelles répondant aux conditions énoncées. Il en existe aussi d'artificielles comme les mines à ciel ouvert qui sont rapidement envahies par l'eau dès que l'on cesse de pomper.

3. En conséquence, dans de nombreuses régions uranifères du Canada, le dépôt des déchets dans des dépressions en milieu sédimentaire signifie le dépôt dans des sédiments lacustres. C'est le sujet du présent rapport.

4. Le dépôt à l'intérieur de sédiments lacustres est un option prometteuse qui pourrait être appliquée au Canada. Elle offre de nombreux avantages, mais présente aussi des inconvénients inhérents.

5. Le dépôt dans les lacs profonds est théoriquement un système passif qui demande et offre très peu de possibilités de modifications ultérieures.

Les avantages et les inconvénients du dépôt des résidus de mines en lac profond sont liés pour la plupart à la nécessité d'atteindre un équilibre acceptable entre la retenue et la dispersion, et se résument brièvement comme suit:

Avantages	Inconvénients
Le dépôt dans un milieu de sédimentation favorise l'isolement avec le temps.	L'augmentation de l'isolement avec le temps signifie qu'il pourrait être plus difficile d'y avoir accès ultérieurement pour une raison quelconque.
La couche d'eau au-dessus des résidus réduit le taux de libération de radon à un niveau négligeable et protège les résidus contre les extrêmes météorologiques.	La couche d'eau peut permettre un transport rapide des contaminants des résidus aux eaux de surface, puis aux eaux en aval du bassin, si ces contaminants atteignent les eaux de surface pendant les périodes de forte décharge ou de circulation des eaux du lac.
La couche d'eau et l'isolement des résidus en profondeur assurent un minimum de bioperturbation à la fois par les communautés humaines et naturelles.	Durant le dépôt des résidus dans le bassin des lacs profonds, il y a des risques de transport des contaminants à l'extérieur du système.
L'infiltration des contaminants des résidus dans le réseau des eaux souterraines sous le bassin lacustre peut entraîner une atténuation géologique en profondeur et un isolement effectif de la biosphère.	La détermination des écoulements dans la roche-mère et de la capacité d'atténuation géochimique est difficile et coûteuse. L'incertitude liée à ces aspects s'ajoute aux incertitudes associées à d'autres variables comme l'hydrologie des eaux superficielles, les réseaux de circulation lacustres, les taux de sédimentation, etc.
Le placement des résidus dans un milieu réducteur (potentiel redox négatif) fait qu'ils se trouvent dans une situation où la plupart des minéraux comme les pyrites ou les arséniures sont stables. De nouveaux minéraux peuvent précipiter ce qui fournit un mécanisme de coprécipitation des contaminants et crée de nouvelles surfaces d'adsorption de ces contaminants.	Le placement des résidus dans un milieu réducteur peut déstabiliser le précipités comme le gypse, le jarosite, les hydroxydes de thorium et de calcium, formés dans le milieu oxydant de l'usine de traitement. Cette déstabilisation pourrait libérer les métaux coprécipités comme le radium, l'uranium et le plomb.

Avantages (suite)	Inconvénients (suite)
Si on choisit un lac profond, les processus naturels du lac peuvent contribuer à créer des conditions telles que la méromicticité. Ces conditions ajoutent une barrière naturelle qui isole les résidus de la biosphère et qui se développe et se stabilise avec le temps.	Les modifications futures du système seront vraisemblablement limitées à une altération des sédiments superficiels ou de la colonne d'eau. Ceci restreindra uniquement les mouvements ascendants des contaminants. Les modifications des migrations vers le bas seront extrêment difficiles, sinon impossibles.

Il y a une série de domaines où nos connaissances sont insuffisantes et où il faudrait des recherches supplémentaires afin de connaître plus complètement les possibilités d'élimination des résidus des mines d'uranium. La capacité à prédire le comportement des contaminants libérés des résidus des mines d'uranium sera améliorée par des recherches dans les domaines suivants:

1. le développement de variables synthétiques combinant plusieurs processus pour donner les équilibres résultants, comme le lieu entre les mécanismes biogéochimiques et le transport physique;

2. l'étude générique de dépôts hypothétiques de résidus dans une variété de systèmes lacustres;

3. l'état chimique et la structure solide des éléments toxiques à divers pH et Eh;

4. l'étude des relations entre la décomposition anaérobique et la dissolution des sulfures et d'autres minéraux accessoires;

5. l'analyse des caractéristiques d'absorption des contaminants sur différentes surfaces, comme l'argile, le matières organiques, le revêtement des fractures, la matrice de roches, etc.;

6. l'étude en solutions pures des conditions chimiques de la coprécipitation des éléments toxiques avec les minéraux formés dans des processus diagénétiques (comme le vivianite);

7. l'étude de l'érosion dans le bassin de drainage en relation avec la sédimentation du lac;

8. des recherches supplémentaires sur les causes et les mécanismes régulateurs de la méromicticité naturelle;

9. la vérification et le développement de modèles mathématiques pour prévoir la stabilité méromictique;

10. le développement de principes unificateurs pour prévoir la bioconcentration, spécialement des principes propres aux conditions chimiques des eaux douces des lacs du Bouclier canadien;

11. la prise de carottes de sédiments lacustres qui ont été en contact avec de l'eau ayant passé sur les résidus auparavant, afin d'étudier les coefficients de distribution et de vérifier les modèles et les théories synthétiques;

12. des études sur le pourtour des lacs (limnocorral) du comportement des contaminants sous divers régimes de Eh et de pH;

13. l'étude des réseaux d'eaux souterraines près des gisements de minerais existants afin de mieux comprendre les facteurs qui influent sur la migration des radionucléides;

14. les prévisions des futures conditions chimiques atmosphériques et de la réponse des systèmes d'élimination des résidus aux divers scénarios environnementaux; et

15. l'étude des limites définies au-delà desquelles il n'est pas raisonnable de s'attendre à ce qu'une évaluation de variables particulières, notamment l'effet du climat, puisse être faite avec certitude.

Number/
Numéro
2

Atomic Energy Control Board
P.O. Box 1046
Ottawa, Canada
K1P 5S9

Commission de contrôle de l'énergie atomique
C.P. 1046
Ottawa, Canada
K1P 5S9

INFO-0070

A QUALITATIVE EVALUATION OF LONG-TERM PROCESSES GOVERNING THE BEHAVIOUR OF URANIUM MILL TAILINGS PLACED IN DEEP LAKES

(ANNEXES)

A study conducted by:

Paul McKee, M.Sc.
Aquatic Biologist

William Snodgrass, Ph.D.
Senior Environmental Systems Engineer

Donald L. Lush, Ph.D.
Manager, Ontario Division

P. Ross, Ph.D.
Limnologist

BEAK CONSULTANTS LTD., Toronto

for the

ATOMIC ENERGY CONTROL BOARD

Contract No.: 34-5-2-15

RESEARCH REPORT

March 1982

ABSTRACT

A Qualitative evaluation of long-term processes governing uranium mill tailings placed in lake sediments.

The emplacement of uranium mill tailings in deep lakes may provide an acceptable method of tailings disposal at certain sites in Canada. Disposal in a depositional environment typical of deep lakes appears to offer greater prospects for long-term stability than present methods of land-based management. From the regulatory point of view, it is necessary to know which factors should be taken into account in assessing the acceptability of such an approach. This report examines the environmental variables governing the behaviour of radionuclides and trace elements in the groundwater systems, lake water, and finally in the biosphere over the short and long-term. Physical, chemical and biological factors are each considered. Conclusions are presented in terms of points for and against disposal in deep lakes.

The opinions expressed in this report are those of the authors and do not necessarily reflect those of the Atomic Energy Control Board.

RÉSUMÉ

Une Méthode acceptable d'évacuation des résidus provenant des usines de concentration d'uranium pourrait consister à les déposer dans des milieux de sédimentation caractéristiques des lacs profonds, en certains endroits au Canada. Comparé aux méthodes usuelles de gestion des résidus en milieu terrestre, ce moyen laisse entrevoir une meilleure stabilité à long terme des résidus. Du point de vue réglementaire, il s'avère cependant nécessaire de connaître les facteurs qui devraient être considérés pour évaluer l'acceptation d'une telle approache. Le rapport considère donc les variables d'ordre environnemental déterminant, à courtre et à longue échéance, le comportement des radionucléides et des éléments en trace dans les eaux souterraines, l'eau des lacs, et enfin dans la biosphère. Tous les facteurs physiques, chimiques et biologiques sont considérés. Les conclusions du rapport portent sur les avantages et les inconvénients de cette méthode d'évacuation.

Les opinions exprimées dans ce rapport sont celles des auteurs et n'engagent pas la Commission de contrôle de l'énergie atomique.

CONTENTS

	Page
0.0 INTRODUCTION AND OVERVIEW	
0.1 Introduction and Overview	I
0.1.1 Mechanisms Controlling Uptake, Release and Transport within Tailings	VI
0.1.2 Contaminant Transport Beyond the Sediment-Tailings System	X
0.1.3 Contaminant Properties in Groundwater	X
0.1.4 Contaminant Properties in Lake Water	XI
0.1.5 Report Organization	XIV
1.0 SEDIMENT-TAILING SYSTEM	1
1.1 U-Mine Tailings in Sediments	4
1.1.1 Biogeochemical Characteristics of Tailings	6
1.1.1.1 Chemical Form of Tailings Upon Deposition	6
1.1.1.2 Geochemical Mechanisms	7
1.1.1.3 Biochemical Mechanisms	10
1.1.1.4 Perspective on Biogeochemical Characteristics	12
1.1.2 Physical Characteristics of Tailings	13
1.1.2.1 Initial Characteristics Upon Deposition	14
1.1.2.2 Porewater Transport Characteristics	14
1.1.2.3 Stability and Cohesiveness	15
1.1.2.4 Solids Transport Processes	15
1.1.2.5 Perspective on Physical Characteristics	16
1.1.3 Perspective on Mine Tailings in Sediments	16
1.1.4 Relationship to Biogeochemical Cycle	17
1.2 Tailings Barrier	18
1.2.1 Physical Characteristics	18
1.2.1.1 Consolidation Processes	18
1.2.1.2 Stability	19
1.2.1.3 Porewater Transport Characteristics	19
1.2.2 Chemical Characteristics	20
1.2.2.1 Uptake Characteristics	20
1.2.2.2 Stability	21
1.2.3 Perspective on Tailings Barrier	21

CONTENTS

	Page
1.3 Lake Sediments	22
1.3.1 Lake Sediment Physics	23
1.3.1.1 Consolidation Processes	23
1.3.1.2 Transport Mechanism Through Pore Water	24
1.3.1.3 Stability	25
1.3.1.4 Accumulation Characteristics	25
1.3.1.5 Perspective on Lake Sediment Physics	26
1.3.2 Lake Sediment Biogeochemistry	27
1.3.2.1 Geochemical Measurements	27
1.3.2.2 Biochemical Processes	33
1.3.2.3 Biological Communities	38
1.3.3 Relative Importance of Transport and Biogeochemical Mechanisms	44
2.0 LAKE WATER BODY	46
2.1 Lake Water Relationships	47
2.1.1 Water Body Physics	47
2.1.1.1 Vertical and Horizontal Lake Characteristics	48
2.1.1.2 Vertical Circulation Characteristics	49
2.1.1.3 Horizontal Circulation Characteristics	54
2.1.1.4 Perspective on Lake Physics	58
2.1.2 Water Body Chemistry	59
2.1.2.1 Chemical Characteristics	59
2.1.2.2 Relevant Biological Growth Factors	61
2.1.2.3 Perspective on Water Body Chemistry	62
2.1.3 Water Body Biology	63
2.1.3.1 Biological Components	63
2.1.3.2 Diversity and Stability of Ecosystems	75
2.1.3.3 Perspective on Water Body Biology	77
2.1.4 Water Body Inorganics	78
2.1.4.1 Physical Characteristics	78
2.1.4.2 Chemical Characteristics	78
2.1.4.3 Perspective on Water Body Inorganics	79
2.1.5 Interactions Between Particulate and Soluble Phases of Water Body	79
2.1.5.1 Biological Production Characteristics	80
2.1.5.2 Biological Decomposition Characteristics	80
2.1.5.3 Ecological Relationship of Community Structure to Water Chemistry	81

CONTENTS

		Page
2.1.5.4	Other Uptake Mechanisms	82
2.1.5.5	Other Dissolution Mechanisms	83
2.1.5.6	Bioaccumulation Factors of Radionuclide Concentrations in Aquatic Organisms	83
2.1.5.7	Perspective on Interactions Between Particulate and Soluble Phases of Water Body	88
2.2	Lake Sediment-Water Interactions	90
2.2.1	Lake Influence on Sediment Properties	90
2.2.2	Net Removal Mechanisms of Trace Metals and Radioisotopes from Lake Water Columns	91
2.2.3	Sediment Water Transport	98
2.2.4	Perspective on Lake Sediment-Water Interactions	101
3.0	HYDROGEOLOGY	103
3.1	Physical Stability of Bedrock	103
3.2	Chemical Stability of Bedrock	106
3.3	Groundwater Flow Characteristics	107
3.4	Biochemistry of Groundwater	109
4.0	INTERACTIONS BETWEEN TAILINGS, LAKE SEDIMENTS AND HYDROGEOLOGY	112
4.1	Tailings, Lake Sediments and Hydrogeology	122
4.1.1	Perspective on Interactions between Tailings, Lake Sediments and Hydrogeology	122
5.0	DRAINAGE BASIN	124
5.1	Terrestrial Characteristics	125
5.1.1	Erosional Nature of Watershed	126
5.1.2	Surface Runoff Potential	127
5.1.3	Chemical Measurements	129
5.1.4	Terrestrial Productivity	130
5.1.5	Perspective on Terrestrial Characteristics	131
5.2	Drainage Basin-Lake Interactions	132
5.2.1	Hydrological Budget Aspects	132
5.2.2	Material Budget Aspects	133
5.2.3	Material Budget Relationships to Biological Regimes	137
5.2.4	Transport Upset Due to Extrema	139
5.2.5	Perspective on Drainage Basin-Lake Interactions	140

CONTENTS

		Page
6.0	ATMOSPHERE	143
6.1	Atmospheric Characteristics	143
6.1.1	Radiation Characteristics	143
6.1.2	Wind Characteristics	145
6.1.3	Precipitation Characteristics	146
6.1.4	Thermal Characteristics	147
6.1.5	Atmospheric Chemistry	147
6.1.6	Prediction of Climatic Change	148
6.1.7	Perspective on Atmospheric Characteristics	150
6.2	Atmospheric Interactions with Drainage Basin	151
6.2.1	Hydrological Budget	151
6.2.2	Material Budget	152
6.2.3	Atmospheric Perturbations on Drainage Basin	154
6.2.4	Perspective on Atmospheric Interactions with Drainage Basin	155
6.3	Atmospheric Interactions with Lake	156
6.3.1	Lake Heat Budget	156
6.3.2	Solar Radiation - Lake Interactions	158
6.3.3	Wind Shear Stress - Lake Current Relationship	159
6.3.4	Water Budget and Mass Budget	165
6.3.5	Perspective on Atmospheric - Lake Interactions	166
6.4	Atmospheric Interactions with Drainage Basin - Lake System	168
6.4.1	Geochemical Weathering Reactions	168
6.4.2	Long-Term Trends in Lake Chemistry	170
6.4.3	Erosional Processes	171
6.4.4	Surface Runoff Characteristics	175
6.4.5	Lake Mixing Characteristics	179
6.4.6	Transport from Lake to Watershed Components	181
6.4.7	Perspective on Atmospheric Interactions	183
7.0	HUMAN INTERACTIONS	185
7.1		
7.1.1	Human Effects upon Atmosphere	187
7.1.2	Human Effects upon Drainage Basin	188
7.1.3	Human Effects on Lake	189
7.1.4	Tailings-Barrier-Sediment System	190

CONTENTS

	Page
7.2 Mitigating Measures	191
7.2.1 Engineering of the Lake Mitigating Measures	192
7.2.2 Environmental Engineering Mitigating Measures	195
7.2.3 Natural and Environmental Controls or Meromixis	196
7.2.3.1 Natural Control of Meromixis	196
7.2.3.2 Engineering Creation of Meromixis	201
7.2.3.3 Perspective on Meromixis	204
7.3 Relationship to Biogeochemical Cycle	204
8.0 DEVELOPMENT OF CRITERIA FOR DEEP LAKE DISPOSAL OF URANIUM MILL TAILINGS	205
8.1 Problem Setting	205
8.2 Problem Definition	207
8.3 Major Variables Selected	208
8.3.1 Description of Major Variables	208
8.3.2 Relative Importance of Major Variables	217
8.4 Summary of all Variables Considered	218
8.5 Illustration of an Approach for Examining the Acceptability of Tailings Disposal in Lake Sediments	239
9.0 RECOMMENDATIONS	253
LITERATURE CITED	256
GLOSSARY OF TERMS	273

CONTENTS

		Page
TABLES		
1.0.1	Observations on geochemical controls of radionuclide transport	8
2.1.1	Some reported values for bioaccumulation factors in aquatic biota containing radioactive isotopes	86
2.1.2	Classification of radionuclides by biological role and periodic grouping	87
4.1	Summary of major biochemical mechanisms and a few associated geochemical mechanisms as a function of Eh	121
6.2	Rainfall chemistry and watershed budget for Hubbard Brook	153
6.4.2	Regional design storm for southern Ontario	177
7.2.1	Geographical and morphometric data for some Canadian meromictic lakes	202
8.1	Selected major variables	209
8.2	Summary of variables and possible measurements of these variables for all hydrological units	219
8.3	Sample analysis of tailings placed in lake	241
FIGURES		
0.1	Sketch of Deep Lake Disposal of Tailings:	
	(a) Initial Deposition	III
	(b) Tailings-Sediment System	IV
	(c) Drainage Basin-Lake System	V
2.1.1	Sketch of various lake-embayment systems	55
4.1	(a) Case I: Sulfide leached from tailings, no organics in tailings	117
	(b) Case II: Sulfide leached from tailings, organics in tailings	118
	(c) Case III: No sulfide leached from tailings but sulfate and nitrate in overlying water	119
	(d) Case IV: Sulfide leached, organics in tailings and iron minerals are present	120
6.3.1	Wind over water as boundary layer problem	162
6.3.2	Regions of water surface drift velocity and results	162
6.3.3	The development of a flow field in a lake	163
6.3.4	Definition-sketch for boundary layer problem of momentum, heat- and mass transfer at a water surface	163
6.4.1	Description of a watershed's weathering reaction	169

0.1 INTRODUCTION AND OVERVIEW

The probable increased use of uranium for energy production and the existing large commercial deposits of uranium ore in various regions of Canada will create large quantities of tailings requiring long-term passive management. Current management practices typically use surface sites. As an alternative, deep-lake passive management is under consideration by various mining companies and regulatory authorities. To aid in these deliberations, it is desirable to develop criteria which may be applied either on an industry-wide or a case-by-case basis. Such criteria will develop through an understanding and determination of the dominant geochemical, biochemical and physical processes (hereafter abbreviated to biogeochemical processes) influencing the tailings, both in the short-term and in the long-term.

The goal of this report is to examine all biogeochemical processes which may influence the long-term behaviour of uranium mill tailings placed in lake sediment and which may need to be considered for establishing criteria. This goal devolves into the following objectives.

1. To analyze all biogeochemical processes which may influence tailings in the sediments of Canadian lakes.
2. To analyze all atmospheric properties and drainage basin processes which may influence the long-term properties of the lake's waters.
3. To develop a comprehensive list of variables which may be useful measures of these various biogeochemical processes and atmospheric-drainage basin properties.
4. To present a short list of major variables which as a minimum should be considered by a proponent or by a regulatory authority for deep-lake disposal.
5. To present an approach for decision-making using these major variables.
6. To make a few statements concerning the adequacy of present knowledge for assessing these major and minor variables.

For the purposes of this document, long-term is defined as being between one century and 10 000 years after placement of tailings. The method of placement of tailings will not be considered except as it affects the properties of tailings one century later.

The concept of disposal of mine tailings within lake sediments offers many potential advantages over conventional terrestrial disposal (for example, see Hawley, 1975). Two significant advantages are the economic desirability of lake disposal over other more costly disposal methods and the long-term depositional nature of the lake environment. These potential advantages are, however, tempered by a variety of uncertainties about both the short-term and long-term behaviour of tailings in lakes. It is also tempered by the lack of proven technology for engineering such disposal so that the environment will not be unacceptably contaminated during the placement of tailins in lake basins. In the face of such uncertainties, a useful approach is to describe qualitatively and quantitatively (where possible) what is known about the geochemical behaviour in such a system, to state the unknowns as best one can and then to determine necessary designs with appropriate safety factors. If it is judged that such designs have too many dangers associated with them in comparison to other methods of disposal, then rejection of the deep lake option may be warranted.

In addressing the impact that tailings have on a lake system, one may approach the problem from a micro-to-macro approach (i.e., tailings outward) or from a macro-to-micro approach (i.e., atmosphere and bedrock systems inward). Aspects of both approaches are suggested in Figure 0.1(a) during and immediately after the placement of the tailings. The micro-to-macro approach is suggested by the illustration of sediment-tailings processes in Figure 0.1(b) because the geochemical properties in the immediate vicinity of tailings are the first barrier of defense against an unacceptably high bleed rate to the ecosystem. On the other hand, the macro-to-micro approach is suggested by the consideration of atmospheric and drainage basin properties of Figure 0.1(c). It is these latter properties which will strongly influence the actual quantity and direction of bleed rates over the long term.

Herein, the decision was made to approach the impact question by considering the pathway for movement out of the tailings through barriers (e.g., sediments, bedrock) into surface waters. This best mirrors the question of concern - what will be the bleed rate (the leaching rate but modified by barriers) from tailings to surface waters. This approach is not completely adequate. Background concentrations of elements in surface waters are a macro-property since background concentrations in lakes have geochemical sources in more macro-systems (e.g., weathering of bedrock and inputs from atmosphere). Some elements which affect the long-range behaviour of tailings will

beak

Figure 0.1: Sketch of tailings placed in example lake

(a) Initial Deposition

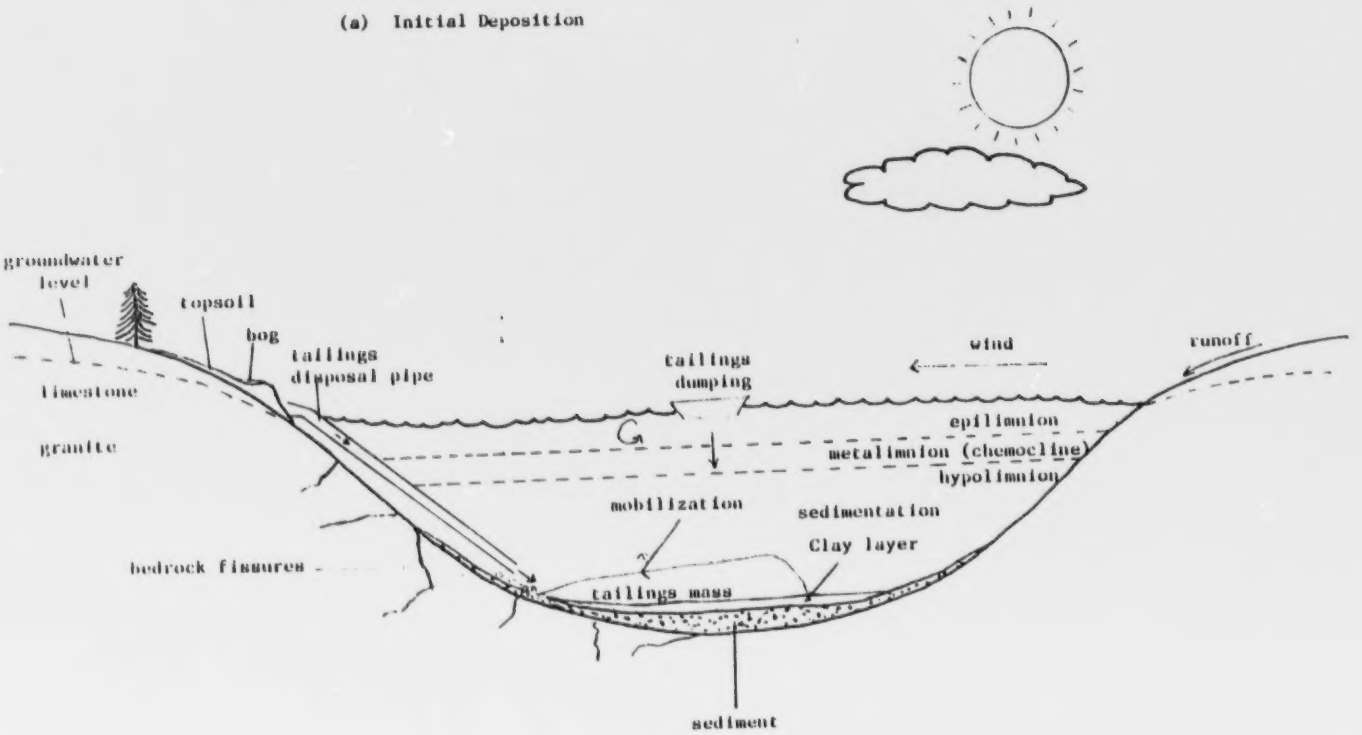


Figure 0.1: Sketch of tailings placed in example lake

(b) Tailings - Sediment system

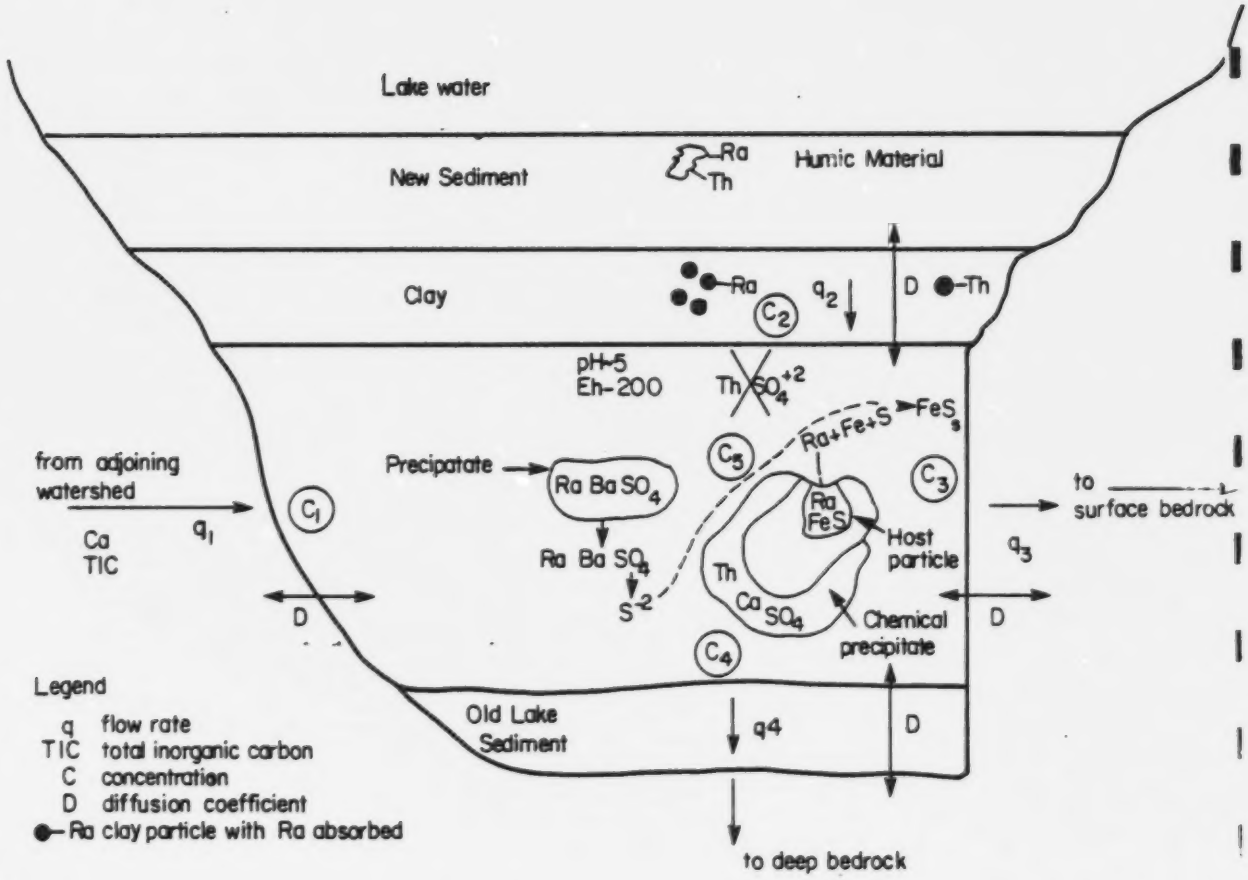
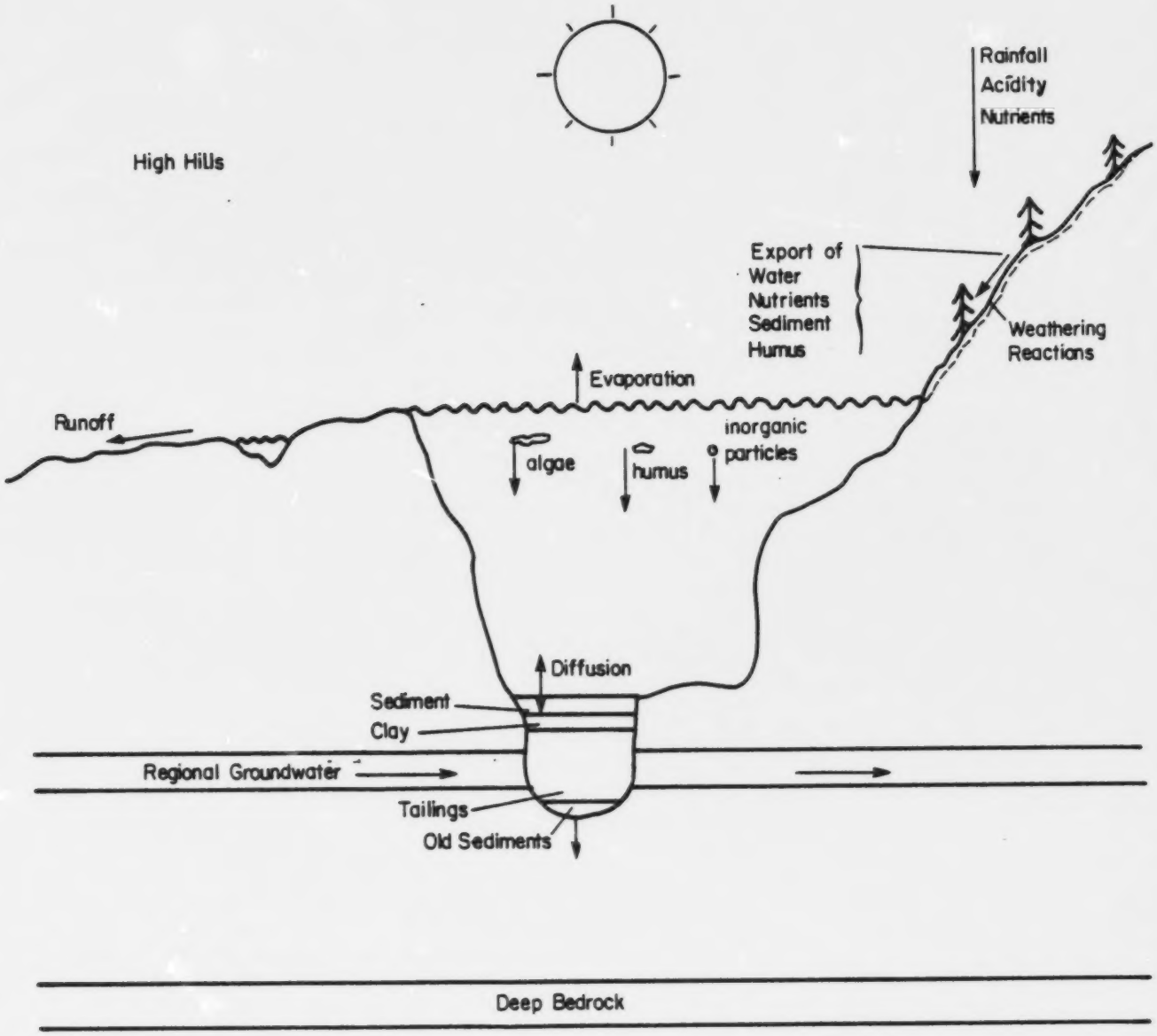


Figure 0.1: Sketch of tailings placed in example lake

(c) Drainage Basin-Lake system



originate in the macro-environment and be carried into a more micro-environment. However, the micro-to-macro approach is advantageous for the question of bleed rate of contaminants out of lake sediments and is therefore retained in this document.

0.1.1 Mechanisms Controlling Uptake, Release and Transport within Tailings

Uranium mill tailings are composed of host ground minerals ranging from sand to clay sized particles, chemical precipitates, and decant liquids. The crushed host minerals have been fractionated from the original ore by processes such as grinding and physical separation. They include, depending upon the host ore characteristics, a spectrum of rock types including sandstones, shales and conglomerates, and minerals such as monzanite, pyrite and arsenide. The actual mix of sedimentary rocks and minerals depends upon the host ore body. The mass of tailings requiring disposal is only slightly less than the quantity of ore mined. While the mass of tailings is similar to the mass of ore mined, its volume typically is not. The blasting and milling of ore resulting in a small particle size adds what is termed a bulking factor related to the increase in porosity of the tailing mass compared to the original ore body. Chemical precipitates typically include gypsum (if acid leaching is employed), calcite, slated lime, iron hydroxides and sulfate sludges.

Contaminants of concern include radioisotopes in the natural uranium (U-238, U-234, Th-230, Ra-226, Rn-222, Pb-210, Po-210) and thorium (Th-232, Ra-228, Th-228) decay series, trace elements (As, Pb, Fe, Mn, Ni) and acid (H^+). Because reduced uranium is very insoluble, most parent ores are thought to have formed under reducing conditions (negative redox potential). Mining of the ore and subsequent milling under moderate to strong oxidizing conditions oxidizes some of the minerals (e.g., uranium) allowing its liberation into solution and subequent recovery. Other accessory minerals (e.g., pyrite) stable under reduced conditions will tend to form new precipitates and minerals stable under oxidized conditions. Milling and chemical leaching liberates many radioisotopes associated with the reduced uranium minerals and recrystallizes them into forms associated with oxic minerals or amorphous precipitates.

When fresh tailings are disposed, over 95 percent of all radioisotopes and stable trace elements are associated with particulate phases. If disposal occurs in an oxic environment, the chemical precipitates may remain stable except for dissolution occurring due to solubility considerations. However, the oxidation of reduced minerals, a reaction catalyzed by bacteria, may lead to acid production and liberation of potentially toxic trace elements such as arsenic. If disposal occurs in an anoxic environment, anoxic and anaerobic reactions will prevent liberation of potentially toxic trace elements associated with reduced minerals (arsenides and pyrite), but may promote the breakdown of oxic-formed precipitates (sulphates, carbonates) and their associated radioisotopes.

In an overview context, then, the first major variable of concern is the group of mechanisms which control the movement of contaminants between the soluble phase (pore water) and the particulate phase in tailings. This variable is useful because of the assumption that the only movement of significance in most lake sediments will be the transport of soluble species. That is, it is usually assumed that tailing particles will remain in the disposal zone unless subjected to erosional influences. For many purposes, description of the relationship between particulate and soluble phases is made using a distribution coefficient. The distribution coefficient is the ratio of concentration in the aqueous phase to the surface concentration on the particulate phase. If a tailings mass has no water flows through it and diffusion is insignificant compared to the rate of sorption-desorption between the soluble and particulate phase, the distribution coefficient completely describes the pore water concentration. This is a steady state concept and it may be thought of as the preference for a particular contaminant to sorb onto the solid phase or to exist in dissolved form.

Over the long term, however, the ability of the surfaces of the tailings particles to sorb soluble materials will change. A more fundamental change will be the potential dissolution of tailings materials, whether due to chemical attack by the pore water or due to bacterial attack (e.g., dissolution of the accessory minerals such as sulfides, and the potential precipitation of minerals caused by changing redox conditions). Such changes require use of concepts of biochemical and geochemical mechanisms in order to explain the changes in relationships between soluble phase and particulate phase concentrations.

Movement of soluble contaminants out of the tailings is the next major process of concern. It consists of advection and diffusion. Advection is the transport of contaminants with moving water, whereas diffusion is transport caused by eddies or molecular bombardment. With or without advection, diffusion always exists.

The physical effects of diffusion in pore water (molecular bombardment of soluble ions by water molecules) is described by a diffusion coefficient. Transport due to diffusion is equal to the product of the diffusion coefficient, the concentration gradient, and the cross-sectional area through which diffusion occurs. The concentration gradient between the tailings pore water and the surrounding water will provide the intrinsic driving force for diffusion of the contaminant from the tailings. The sediment porosity and degree of interconnection of void spaces will affect the "area" through which diffusion occurs. In addition to pore water diffusion, diffusion within the solid tailings particles also occurs. Solid diffusion enables contaminants contained in the middle of a tailings particle to migrate to its exterior and be available to pore water. This process will probably influence the interstitial water concentrations only in the long term, after dissolution of the same contaminant from sites on the particle surfaces has ceased to be important. The change of dominant mechanism for supply a contaminant to pore water from dissolution to solid state diffusion is another example of a change of geochemical mechanisms.

If the bulk hydraulic conductivity of the tailings mass can be engineered to less than 10^{-6} to 10^{-7} cm/sec in all directions, then the bulk diffusion coefficient and mechanisms controlling the distribution coefficient are expected to be the dominant variables regulating the rate of release of contaminants in most tailings to the surrounding environment. Diffusion transport of contaminants from the tailings mass will change with time as the magnitude of the diffusion gradient changes across the tailings mass and as the concentration of solutes, ionic strength, and redox potential in pore water change. The Eh and pH changes will be controlled through sulphide diffusion outward, oxygen and carbonate diffusion into the tailings, and degradation of organic matter in the tailings.

If the hydraulic conductivity of the tailings mass exceeds 10^{-6} to 10^{-7} cm/sec, advective transport of contaminants from the tailings mass can dominate. Under these conditions, factors external to the tailings mass itself, including hydrological and chemical factors, will begin to influence the flux of contaminants from the tailings.

A major factor influencing advective transport (movement of water through voids) is the hydraulic gradient which exists across the tailings mass. The hydraulic gradient is the slope of the piezometric surface of water in the tailings

The flow rate of water through the tailings mass (which is a function of the product of the hydraulic conductivity and hydraulic gradient) will be superimposed upon the diffusional transport. The higher the hydraulic conductivity and hydraulic gradient, the less important is diffusional transport. The advective flow through the tailings mass and hence contaminant flux can be regulated to some degree by the natural or engineered confining barriers surrounding the tailings.

If tailings are to be placed within a lake basin, it is likely that they will be placed upon a naturally-occurring, fine-grained lake sediment. Alternatively, some unconsolidated material such as clay may be placed between the tailing mass and the natural lake bottom. Any underlying unconsolidated layer and any overlying unconsolidated layers, whether engineered or naturally deposited, may be regarded as confining barriers. These barriers will affect the rate and timing of loss of contaminants from the tailing mass by i) having several solid-liquid distribution coefficients for contaminants which may differ from those of the tailings mass, ii) providing a potentially lower hydraulic conductivity and diffusion coefficient than found within the tailings mass, and iii) providing a wider zone across which the diffusion gradient is exerted and hydraulic head is dissipated.

For sediments, it is useful to use the concept of distribution coefficient if only inorganic material is present; if organic particulates are present, an additional distribution coefficient is needed. If minerals such as vivianite form (e.g., Nriagu and Dell, 1974), then a third distribution coefficient which increases in importance with time may be used. If the ratio of organic matter to inorganic matter and the mixture of adsorption and precipitation mechanisms change over time, consideration of the change of biochemical and geochemical mechanisms becomes necessary.

All of these factors can be engineered in a chemical or physical sense to restrict contaminant loss rates, if a preliminary evaluation of contaminant behaviour without confining barriers indicates that further restrictions on contaminant transport are required. It must also be recognized that as in the case of the tailing mass itself, the physical and chemical characteristics of this confining barrier may change with time. An example of this would be the diffusion of high-sulphate water into the confining layer. This will change the ionic

strength and possibly pH. Evidence is also available which shows that waters high in calcium sulphate can alter the structure of sodium-saturated bentonitic clays, possibly through partial calcium substitution for sodium, therby altering physical properties such as diffusion coefficients and hydraulic conductivities.

0.1.2 Contaminant Transport Beyond the Sediment-Tailings System

Once contaminants have moved beyond these natural or engineered barriers, a flux of contaminants may be expected into the underlying till and bedrock groundwater, and into the overlying water column. In the absence of any hydraulic gradient, diffusional losses will be principally dependent upon the diffusion coefficient and concentration differences between the pore water and underlying groundwater or between the pore water and lake water. Where the hydraulic gradient and hydraulic conductivities are sufficient to allow advective flow to be the dominant mode of transport, a directional contaminant plume will exist.

0.1.3 Contaminant Properties in Groundwater

If the dominant plume direction is downward into underlying till or bedrock, it can be modelled using either porous media or fracture flow models or some appropriate hybrid. This enables the prediction of the direction and the rate of movement of contaminants. If transport occurs to the biosphere, a model can be used to predict the timing and the rate of release of contaminants from a groundwater flow system to the biosphere. Movement of this contaminant plume through porous media will depend upon many of the same factors discussed previously for confining layers. In addition, the effects of dilution and diffusion become important influences upon the concentrations released to the biosphere.

If local hydraulic gradients or diffusivity transport are vertically upward, a contaminant plume will exist towards the overlying lake water. This can be modelled in a manner similar to a downward plume.

The movement of a contaminant plume through a fractured medium becomes somewhat more complex. Net water and contaminant velocities may be considerably higher; the determination of dilution and dispersion becomes somewhat more uncertain. As with porous media, the distribution coefficient is important, but this is complicated by the fact that surface reactions will occur i) along fracture surfaces, and

ii) within the rock matrix surrounding the fracture. The rate of reaction within the rock matrix is controlled, however, by the contaminant diffusion coefficient within the rock matrix and by the concentration difference within the rock matrix. This concentration difference is the difference between the in-fracture concentration and the concentration on the solid surface. In-fracture gradients and flow velocities are influenced by the fracture aperture size. Depending upon the relative influences of i) the distance of fracture flow before emergence to surface water, ii) the rate of advective flow, iii) the dilution and dispersion involved along the flow path, and iv) the influence of the solid-solute distribution coefficient on contaminant retardation, the plume may or may not be detectable upon discharge to the surface water system at any given point in time.

0.1.4 Contaminant Properties in Lake Water

Within the lake there are physical, chemical and biological processes which are responsible for the dispersion of contaminants into and removal of contaminants from the biosphere. For convenience, these processes are discussed separately but in nature form an integrated system.

Consider a simple system in which the only significant route for transport of soluble materials from the tailings to the lake water is groundwater and in which the processes of advective (convective) transport and diffusion are important. No surface inflow or outlet and no interactions with water column particulates are significant. In this instance, contaminants diffusing or being advectively transported across the sediment confining layer would be diluted and dispersed within the lake. The principal mechanism for lake water dispersion and dilution would be wind and thermally driven currents. The relative magnitude of these processes would depend on season and also depend upon lake basin morphometry and local climate. In the absence of any inflow or outflow, the concentration would continue to build as long as the tailing mass constituted the only source. Net evaporation would equal the rate of groundwater inflow. At some point in the future, a steady state concentration would be reached in the lake; it is conceivable that the lake concentration would be higher than the pore water concentration due to the upward advective flow.

In most lakes, however, there will be inflow and outflow either through surface or groundwater flow systems. Surface water flow will transport a certain amount of material downstream. Once a steady state condition has been achieved, loss from the tailings mass and

overlying sediments to the overlying water column will equal the export to the downstream portion of the watershed, again assuming that the soluble contaminants are unaffected by biochemical reactions.

Any consideration of tailings management must also include the chemical factors influencing contaminant transport. One of the most basic of lake processes which may be affected by chemistry is that of turnover. Most lakes mix top to bottom due to wind driven currents at certain times of the year; this is called turnover. Some lakes, because of very strong chemical gradients existing in their deep basins, develop a chemocline or zone of rapid chemical change. There is a sharp density difference across the chemocline which resists vertical advective transport of water. Accordingly, transport across the zone below the chemocline (the monimolimnion) will be diffusion controlled. In most inland situations, long-term maintenance of a chemocline is dependant upon biological processes occurring within the lake or upon inflow of groundwater rich in salts.

Throughout the water column, there are found a variety of suspended particulate materials. These are both mineral and biological in nature and have contaminant distribution coefficients associated with them. The net effect of these materials is to chemically bind contaminants from the surrounding water until a steady state equilibrium is attained. In the case of mineral particles and non-living organics (e.g., various clays and humic materials), the sorption process is passive and relatively predictable. When living biological particulates are involved, the uptake processes are active. The net result of both passive and active sorption is that contaminants are stripped (or sequestered) from the water column by the particulates and then removed through settling to the sediments of the lake. The particulates come from land-based erosion and from biological growth within the lake. In addition, shallow sediments within the lake basin are continually being resuspended by wind driven currents and resettled in deeper areas of the lake with an overall net transport of fine sedimentary particulates, with high surface area to mass ratios, toward the deeper lake sedimentary basins.

The partitioning of contaminants between the liquid water phase and the solid particulate phase and the subsequent settling rate of these particulates are the principle mechanisms responsible for controlling the steady state concentration of contaminants in the water column over the long term. That is, the quantity and quality of particulates in the water column control the rate of removal of contaminants in the water column. This stripping of contaminants from the water column by settling particulates will also affect diffusion gradients across the

monimolimnion. The long term buildup of sediments will also increase the distance across which the diffusional driving force is dissipated within the sediments.

The biological systems of the watershed and, more specifically, of the lake, are strong controllers of physical and chemical processes generally considered responsible for tailings contaminant containment or dispersion. Processes which occur in the terrestrial portion of the watershed and in the streams feeding into the lake will control the properties of many particles in the water column.

Biological processes within the watershed first begin in the headwater where terrestrial productivity dictates, to a large degree, the amount of both mineral and organic detritus washing into lakes. Terrestrial productivity modifies the basic weathering reactions and tendencies for erosion. This mineral and organic particulate material is responsible for stripping contaminants from the water column if they enter the lake. In addition to particulate inputs, many soluble nutrients and refractory organics enter the lake. Refractory soluble organics such as humic and fulvic acids have the ability to complex with contaminants and thus compete with particulates. While the particulate organics strip contaminants from the water column, the soluble materials stabilize the contaminants in solution, thereby possibly increasing their bioavailability and the probability of downstream transport.

The partitioning of contaminants between the soluble and particulate pool strongly influences the long term equilibrium level which will eventually be attained within the lake and downstream.

It should be recognized that complexing with soluble organics is not necessarily detrimental, as much of this material is effectively unavailable for biological uptake and bioconcentration. It does, however, increase the net level of contaminants in the water column and thus, in the case of radionuclides, the net immersion radiation dose in water.

The inorganic soluble component (principally classified as nutrients) coming into the lake from the surrounding watershed influences to a large degree the lake's rate of biological production. The rate of biological productivity influences both active and passive transport to the sediment. This net transport is composed of biomobilization of certain elements from sediments and particulate sedimentation of others. Since biomobilization is strictly an active process, it tends to be more selective than particulate settling. Biomobilization from

lake sediments can vary both spatially and seasonally. It generally occurs more in summer and in shallow waters, especially in macrophyte beds. Little occurs in winter or below a chemocline.

Because biological uptake is an active process it can be both more selective and have a higher effective distribution coefficient than uptake by mineral or organic detritus. Biological processes are important because of the following:

1. Living particulates from algae to fish can actively maintain contaminants in the water column on a seasonal basis and hence, affect seasonal downstream transport.

2. The biological system is dynamic, responding to a whole set of climatic and watershed variables which may change through time.

3. Long and short-term successional cycles may affect different contaminants to different degrees.

4. The decay of dead biological systems below a chemocline releases solutes which maintain density gradients and, hence, influence physical transport processes deep within the lake basin.

5. Biological particulates within the upper layers of the water column absorb light and hence influence the heat budget of a lake and thermally-induced circulation.

6. Biological particulates can be responsible for the greatest export of contaminants from the system and to specific target organisms (i.e., fish to man).

7. In many Canadian Shield Lakes, biological particulates are the major contributor to lake sediment buildup.

0.1.5 Report Organization

The following sections of this report will expand upon the physical, chemical and biological processes involved within the watershed and the manner in which they interrelate and affect contaminant disperison from a tailing mass in the bottom of a generic lake system. The basic building blocks used are hydrological units: atmosphere, watershed, lake water, lake sediments, and groundwater.

For examination of the migration of contaminants, this analysis follows from micro to macro environments. Hence, tailings, then tailings barrier, then lake sediments are considered. Next, the properties of lake water (physics, chemistry, biology) and the properties of hydrogeology are considered. Then, interactions between the lake water-sediment system and the groundwater-lake water-lake sediment are considered. Interactions are defined as the transfer of mass or energy between hydrological units.

For examination of the background concentration of contaminants and of other constituents in the lake water, this analysis follows from relatively intermediate to more macro hydrological units. First, the properties of the drainage basin are considered. Then drainage basin-lake interactions are examined. Lastly, atmospheric properties and the interaction of the atmosphere with the different hydrologic units (drainage basin, lake, hydrogeology) are considered. Consideration of the drainage basin-lake system as one unit is useful because the lake concentration of many constituents is dependent upon watershed properties. However, for other materials over the long term, atmospheric influences on the drainage basin-lake system and upon groundwater are dominant. For such considerations, the atmosphere-drainage basin-lake must be considered as one unit.

The report is therefore organized to consider the properties of each hydrological unit and then the interactions between adjoining units or groups of hydrological units. Interactions between hydrologic units are transfers of energy (mechanical energy, heat energy) and transfers of mass (e.g., radioisotopes or organics). The properties of a hydrological unit are separated (organizationally) from the interactions between hydrological units because the majority of environmental measurements are made on each hydrological unit. These are estimates of one property at a point in time (e.g., pH of rainwater, concentration of algae present in lake water, pore water concentration of radium in the lake sediments, sulphide concentration in the tailings, soil moisture concentration, bedrock porosity or an aspect which influences transport through the unit). Only a few of the interactions between hydrological units are routinely measured (e.g., solar radiation, rainfall, stream flow rate), despite the fact that it is these interactions which determine variables of concern to this project such as the bleed rate of contaminants to the lake water.

For management purposes, one desires to know a few of the properties of each hydrological unit (e.g., radioisotope concentration in the lake water) and a few of the interactions (e.g., bleed rate of

radioisotopes from the lake sediments to the lake water). These two aspects (e.g., concentration and transport rates) cannot be related except by the formulation of the proper mathematical relationships. For certain properties and interactions, one needs to know both; for others, it is sufficient to know one or the other. In this report, attempts are made to highlight those which will be most useful for the majority of situations.

1.0 SEDIMENT-TAILING SYSTEM

The decision of whether or not the placing of uranium mine tailings within lake sediments is environmentally acceptable requires a knowledge of the total contaminant inventory and a knowledge of the rate at which any given contaminant may be released to the open biosphere. In the biosphere, impacts upon man and other biological species may result.

The concept of a contaminant inventory can be broken into three conceptual classes. They are as follows:

1. Contaminants which are readily mobile. In most instances, this includes contaminants dissolved in pore water and contaminants which could be leached from tailings by a leaching water of pH 4.0 if the pore water volume is displaced four to five times. Conceptually, this would be analogous to leaching with acidified lake water or a low-pH groundwater.

2. Contaminants which have an intermediate degree of either physical or chemical mobility. From an operational chemical perspective, this would describe tailings materials which are soluble and which can be leached from a tailings sample at a pH of 1.5 (e.g., 0.05 M HCl). This is analogous to pH levels which may be achieved if iron-oxidizing bacteria were present in sulphide-rich tailings. This chemical test would describe biochemical or geochemical processes which would affect neutralization precipitates formed in the uranium mill on the surface of the tailings, and chemical sludges such as the $RaBaSO_4$ sludge that may be mixed in with the tailings. It is possible that under severe conditions of very steep hydraulic gradients and channelization of flow within the tailings mass, this material may escape in a fine particulate or colloidal form.

3. Contaminants that remain within the native mineral component of the tailings (i.e., in the host material). These are generally the coarse fractions of the tailings and those fines which were not fully dissolved during the acid or base leaching process. The contaminants within these mineral particles may be considered refractory in nature since they have survived severe acid or basic leaching. Their loss to interstitial water and hence to the open environment will likely be very slow and rate-limited by solid-state diffusion within the mineral particle itself.

It is possible that any type of mineral contaminant may fall into any one of these conceptual classes. It is also possible for contaminants to change form chemically within the tailing mass. Where there is a potential chemical gradient present, a chemical change is inevitable. Relevant examples include a fully-oxidized sulphate compound in a negative redox environment and, conversely, a fully-reduced sulphide in an oxidizing environment. The former arises from transport of sulfate into a sediment system and the latter from S^{-2} migration out of sediment into lake water. If oxidizing or reducing actions are strictly of a chemical nature, the change will be relatively slow causing a slow increase or decrease in the release rate of contaminants depending, upon the resultant solubility of the products and the distribution coefficients associated with any new solid phases produced.

If there is a carbon and nutrient source present, it is likely that a microbial (bacterial) population will either be present or develop to exploit the carbon and nutrient source. The net effect of the resultant biochemical reactions is an acceleration of chemical reactions and a more rapid release of contaminants into solution. If a favourable chemical gradient is present, this will result in the potential for an increased rate of transport of the contaminant from the tailings mass.

In addition to a catalytical ability, bacteria can also act to remove bacterial toxins from their local environment. There are many known instances in nature where microbial populations, in order to "clean up" their own microenvironment of toxic materials (e.g., Hg, Pb, Sn, Se, etc.), will methylate the offending contaminant. This creates either i) a water soluble product which is removed from their microenvironment by the flushing action of water, or ii) dissolved gases which enter into gas bubbles if methane or nitrogen bubbles are formed. In addition to methylation reactions, soil and sediment microbes are known to produce carboxylated polysaccharides which may be present either in particulate or soluble form. These materials have very pronounced cation exchange properties and may strip weakly bound contaminants from solid surfaces and serve as a transport medium for contaminants to and through the open environment. The consequences of these materials to mans use of waters may be restrictive if levels are too high. At low release rates these processes are of ecological importance as many contaminants at low levels are essential trace elements. These microbial dispersion mechanisms may play a vital role in the aquatic ecosystem in preventing trace element deficiencies from developing. The key point to remember is that we are dealing in many cases between the extremes of elemental deficiency or insignificant ecological effects and

elemental toxicity with measureable ecological effects. Any extention above or below this optimum range can have direct or indirect effects on man.

In order to determine whether or not microbial reactions within the tailings mass will be significant in the future it is necessary to assess the carbon supply present in the tailings mass and the levels of necessary macro- and micro-nutrients, particularly nitrogen and phosphorus. If nutrient conditions are favourable and chemical gradients exist, microbial activity cannot be ignored and it should be considered as a significant mobilization mechanism affecting the mobility of all three general groups of contaminants. It should also be noted that microbially mobilized contaminants are in all likelihood complexed with microbially produced extracellular products. These organic complexes may also display an enhanced degree of mobility over inorganic contaminant species. These organically complexed contaminants of microbial origin should not be confused with inorganic contaminants released under acid conditions which may result from bacterial oxidation of sulphide minerals under positive redox conditions.

Differentiation of Geochemical from Biochemical Mechanisms

Geochemical processes include those processes which control the long-term formation of minerals and mineral properties. Geochemical processes are defined herein as those long-term processes which influence the distribution of individual elements in the various parts of the earth (e.g., atmosphere, hydrosphere, crust), in minerals, in rock, and between lake sediments and pore water. It is defined to exclude transport processes such as diffusion, advection, consolidation and plate uplift and to exclude biochemical processes, except as they control the pore water chemistry associated with deep sediment areas. Biochemical processes are defined as processes driven by or mediated by organisms (bacteria, algae, yeast, etc.). These processes speed up or slow down geochemical processes.

Another useful differentiation is that geochemistry examines geological processes from a chemical perspective while biochemistry examines biological processes from a chemical perspective. Hence, geochemical processes are those which occur over the long term while biochemical processes are those which occur over the short term.

Geochemical processes may be viewed from the point of view of chemical thermodynamics, while biochemical processes may be viewed from the point of view of chemical kinetics. Where a pseudo-equilibrium point is reached in biochemical processes, the system is at steady-state. Where a pseudo-equilibrium point is reached in geochemical processes, the system is at thermodynamic equilibrium.

Over some time-frame, biochemical and geochemical processes become one and the same. For example, the diagenetic formation of vivianite, (the crystallization of FeS from Fe^{2+} in pore water), is a geochemical process which is strongly influenced by biochemical processes; the reducing conditions necessary to produce Fe^{2+} and S^{2-} are caused by anaerobic bacterial activity. For such a process, both descriptors (geochemial, biochemical) can be used. For our purposes, we will try to use the perspective of whether we are talking about the long-term formation of a mineral or about biological conditions causing the anaerobic conditions as a way of differentiating geochemical from biochemical. For examination of surficial pore water concentrations of sediments, we will again try to keep a clear definition.

1.1 U-MINE TAILINGS IN SEDIMENTS

Over the short-term, it is useful to think of the movement of leachates from tailings as the transport of soluble materials with water flowing through a black-box. This assumes that the particulate phase does not move. Accordingly, calculations of contaminant migration may be described in terms of pore water concentrations and three mechanisms controlling these concentrations: i) advective transport (physical movement of water in one direction, e.g., that described by permeability, using Darcy's law to relate velocity of flow to the resistance to flow (1/permeability coefficient) and the hydraulic gradient), ii) diffusivity-type transport (e.g., molecular, due to elemental bombardment by water molecules, or eddy, due to turbulence), and iii) a chemical distribution coefficient (relating the pore water concentration to the surface concentration on the particles, and incorporating geometry effects such as surface area of particles to volume of pore water, precipitation-dissolution reactions, adsorption-desorption reactions and pore water complexation). If significant biochemical decay occurs, a fourth term is necessary. Examples of biochemical decay include radioactive decay of a radioactive isotope and bacteriological decay of organics. (Note: the term biochemical decay as used here means the change of the state of a substance from one chemical state to another. It can be caused by strictly chemical changes or by organisms.) For transport of radium through clays, the fourth term becomes significant for low flow rates.

This simple differentiation of the major phenomena (advection, diffusion, chemical distribution coefficient) controlling contaminant migration provides a building block for conceptualizing the long-term process. Unfortunately, it is insufficient for every situation and and every material because of the interactions between organic matter and biota and the leaching of chemicals. The distribution coefficient is essentially a steady relationship between the pore water concentration of an element and the surface concentration. There can be unpredictable shifts over time in the biochemical or geochemical system controlling the "distribution" coefficient; this limits the usefulness of the "black-box" approach. If surface properties change or the biogeochemical system shifts, one must use available knowledge to examine these changes in order to predict the non-steady-state behaviour of the interrelationships between constituents in pore water and in the solid phase over a 10 000 year time frame. In the extreme, a contaminant may completely leach from the tailings mass and meaning that a distribution coefficient does not exist. For such a substance, the actual mass in the tailings (solid plus liquid) is the necessary quantity which must be examined, rather than its pore water concentration. Such materials potentially include components of neutralized products and chemical sludges.

For examining tailings and other materials in lake sediment systems, it is useful to divide them into soluble (dissolved) aquatic forms and particulate forms (attached to or incorporated into solid particles). For pore water, relevant master variables at a point in time and in space are the pH, Eh, ionic strength, major elemental composition, contaminant concentration level, volume of pore water per unit volume of tailings, and the biochemical and geochemical mechanisms controlling the relationship of these elements between the soluble and particulate phases. For solid phases, the relevant master variables are the surface area exposed to pore water and the biochemical and geochemical mechanisms. The pore water volume and particle surface area may be combined to form parameters such as a mean pore water length scale or porosity.

For describing the physics controlling the movement of contaminants through porewater, relevant master variables are the permeability coefficient and the diffusivity coefficient. For the movement of particulate phases, relevant master variables are consolidation rates, rate of burying of sediments, advective movements (e.g., lifting due to erosion, hydrothermal springs, or earthquakes) and solid state diffusion (of contaminants from the inside of to the outside of particles of tailings particles or precipitates).

These two aspects are now considered under biogeochemical characteristics and physical characteristics of the tailings mass.

1.1.1 Biogeochemical Characteristics of Tailings

1.1.1.1 Chemical form of tailings upon deposition

Possible Measurements

1. percent of each contaminant in different forms
2. rate of breakdown or dissolution of particulate forms as a function of different particle size

It is useful to divide tailings into different forms to allow inferences about their environmental behaviour upon injection into the sediment. One system of division is according to their chemical leachability or to their chemical forms upon the completion of mining-milling. Such a system might divide the U-mill waste into three forms: i) the host material (parent ore after leaching), ii) the chemical neutralization precipitates on the host material, and iii) chemical sludge (e.g., a $RaBaSO_4$ co-precipitate formed by chemical precipitation, coagulation and settling). This differentiation reflects the fact that a chemical sludge will typically have a higher concentration of radium than the host material after mining-milling. Such a differentiation may not be as useful for contaminants which are distributed only in one phase or for mill waste in which the chemical sludge is a very small fraction and is distributed evenly through the tailings such that its properties are not readily discernible from the bulk mass.

Knowledge of the different forms of tailings and their leachability is essential for making estimates of the chemical form of the tailings 100 years after deposition and for making estimates of mechanisms which will control any long-term dissolution reactions. Knowledge of these forms permit estimates of the rate of solid-state diffusion through the solid particles to pore water.

Specific leaching experiments may be made upon the tailings to gain insight into their probable short-term and long-term breakdown. It is necessary to assess breakdown over the short-term in order to permit estimation of their characterisitcs at the beginning of a long-term period. Knowledge of leaching processes and other mining-milling processes permit development of an appropriate set of laboratory leaching experiments to simulate geochemical and biochemical mechanisms which may control the long-term picture. In particular, the chemical or biological extractability of contaminants from the three major forms

of tailings should be considered. Due to the demonstrated characteristics of different size fractions of tailings, experiments which examine the preferential breakdown patterns of major chemical constitutents as a function of particle size distribution for the major forms of tailings would be of great value. In particular, constituents which are health hazards (e.g., such radionuclides as Ra, U, Th) or which may control the speciation chemistry and geochemical mechanisms should be examined. Examples of the latter are assessory minerals containing sulphides (and their potential control on Eh), and precipitates containing carbonates (and their potential control on pH through the release of Ca and carbonate). Experiments designed to evaluate the biochemical oxidation of accessory minerals and organics, if present, (e.g., potential C, N and P sources) would also be valuable.

The extent of these experiments and methodologies may be new to professionals involved in management of tailings; many of these experiments will have been conducted in one form or another by other disciplines (e.g., geochemistry, chemistry, chemical sedimentology, environmental engineering, biochemistry, and chemical engineering). Knowledge from these other fields should aid in avoiding past mistakes made in these studies. Perhaps a pilot study which examines the experiments of these other fields with the aid of defining possible experiments which need to be conducted by a U-mining proponent may be in order. Of course, no pilot study can define all anticipated problems; hence, it cannot be looked at as a general answer to all questions but rather as an effort to evaluate general experimental methodologies with the specific properties of tailings in mind.

1.1.1.2 Geochemical Mechanisms

Possible Measurements

1. pore water concentrations
2. crystalline forms
3. uptake-dissolution mechanisms
4. temperature

Knowledge of geochemical forms of tailings and their interactions with pore water permit an assessment of the relative long-term mobility of tailings contaminants. Inferences about this behaviour are summarized in Table 1.0.1.

As indicated above, certain master variables exist for describing pore water and its relationships with solid phases. The pH influences the

Table 1.0.1: Observations on geochemical controls of radionuclide transport

I Tailings - Radium

1. Radium does not form discrete minerals but coprecipitates with a variety of minerals.

2. Radium can adsorb onto crystals (e.g., $BaSO_4$), clay minerals, manganous oxides and organic matter.

3. Of all alkaline earth series it would tend to remain preferentially adsorbed.

4. It has the least tendency of all alkaline earths to form complex ions.

5. Ra in U-ore is highly soluble in HCl and HNO_3 but not H_2SO_4. This suggests that H_2SO_4 leaching does not remove as much Ra from the tailings as is possible.

6. Ra in acid-processed tailings is leachable with distilled water (Comment: this is probably an aerobic test.)

7. In slimes, Ra may be sorbed onto active sites of silicate minerals (a surface adsorption-mechanism) as well as coprecipitated with several alkaline earth sulfate compounds (e.g., $CaSO_4$, $SrSO_4$, and $BaSO_4$) of widely differing solubility. (These represent multi-component solubility considerations.)

8. In chemically neutralized precipitates, adsorption is important but more Ra may be in a coprecipitate.

9. Some data exists for Ra mobility in oxidized environment (e.g., leaching of FeS minerals by T. ferrooxidans), but man's understanding of its mobility in anaerobic environments appears to be relatively sparse.

10. High radium concentrations in groundwater around ore bodies suggest that it can mobilize Ra from the precursor of tailings. (This is assumed to be a reducing zone.)

11. Ra exhibits only the (2+) oxidation state in solution.

Table 1.0.1: (continued)

II Tailings - Thorium

1. Th exhibits only the (4+) oxidation state in solution.
2. Above pH 2-3, it undergoes hydrolysis leading to extensive sorption by clay minerals and humic acid.
3. Th precipitates as highly insoluble hydroxide or hydrated oxide (Ksp of $Th(OH)_4 = 10^{-39}$) above pH range 7.-8.
4. Th readily forms complexes with organics, F^-, SO^{-2}, PO^{-3}, Cl^- and NO_3, leading to its transport mainly as a colloid or as a soluble anionic complex.
5. Groundwater with low pH, low calcium, low ionic strength and high sulfate concentrations favours the transport of Th, but similar water inhibits Ra transport.
6. Acid-leaching during milling may dissolve 30% to 90% of Th in ore; neutralization will probably result in precipitation or coprecipitation of Th.

III Tailings - Uranium

1. U^{4+} (uranous) and U^{6+} (uranyl) states are stable in geological environments. The transition occurs within normal range of geological environments.

$$U^{4+} + 2H_2O = UO^{2+} + 4H^+ + 2\,e^- \quad (E^\circ = +0.27V)$$

2. Transport generally occurs in oxidized waters with UO^{2+} or U^{6+} complexes
   a) in oxidized acidic waters, UF complexes dominate
   b) in neutral and alkaline waters, phosphate and carbonate complexes dominate
   c) hydroxyl-, silicate-, organic-, and sulfate-complexes may be important in particular regimes
   d) soils, humic acid, fulvic acid, coal, iron oxides, clays adsorb U reversibly. For it to be fixed, it must be reduced to U^{4+} (by substrate or H_2S).

form of many ions and their dissolution characteristics. The Eh is a measure of whether an oxidized or a reduced form of an ion will be present.

The ionic strength affects the chemical activity of soluble ions by affecting their ability to form precipitates because it influences their ability to migrate through water. The ionic strength is a function of all ions present. However, over time the concentration of ions will change; hence ionic strength contribution to concentration due to the mass balance transport and uptake-dissolution geochemical mechanisms (and biochemical mechanisms) will control the master variable.

In many freshwaters pH is controlled by the carbonate buffering system (see section 2.1.2 for more details). Accordingly, the pore water pH will depend upon whether this buffering system continues to control it or whether a sulfide or organic acid also contributes to buffering. This will depend upon the relative ability of different dissolution-uptake reactions to supply sufficient mass of either material (carbonate, sulfide, organic acid) to cause it to control the pH. Similar arguments may be presented for Eh (see next section), ionic strength, etc.

Crystalline forms describe the minerals which are present and which may dissolve to affect pore water concentrations. Crystalline forms also describe the possible minerals which may form (precipitate) if the pore water is supersaturated with respect to this mineral.

1.1.1.3 Biochemical Mechanisms

Possible Measurements

1. composition of organic matter
2. probable biological communities
3. possible chemical leaching agents
4. temperature

If particulate or soluble organic carbon and sufficient nutrients (N, P) and micronutrients are present, bacterial communities will probably develop. These communities are reviewed in depth in sections 1.3.2, 2.2, and 4.0. The essential point of interest is that organic matter is the food (energy) source for bacterial growth, and an oxidized compound is the final electron acceptor. Hence, oxygen is the final electron acceptor for aerobic bacteria. Nitrate is the final electron acceptor for denitrifiers (nitrate being transformed to nitrogen gas),

and sulfate is the final electron acceptor for sulfate-reducers (SO_4 being transformed to S^{-2}) in anoxic and anaerobic regimes, respectively.

The development of anaerobic activities occurs if all oxygen and nitrate present has been utilized such that SO_4^{2-}, CO_2 or organics are the only remaining materials capable of acting as "final electron acceptors". Such characteristics cause reduced conditions, making species such as S^{2-}, Fe^{2+}, Mn^{2+}, NH^3CH the stable species. The ability to develop such conditions depends upon having no oxygen penetration to these areas and the presence of sufficient organic matter. The types of biological communities which develop are a function of biological competition, type of organics, availability of electron acceptors, etc.

For our purposes, a most useful approach is an analysis of major bacterial groups (e.g., sulfur reducers, etc.) in conjunction with the analyses of the composition of organic matter and the chemical electron receptors found in the tailings. Then one may view the problem at hand as one in which transformation occur between chemical states because of biological activity. Further, bacteria may be viewed as agents acting as catalysts of geochemical reactions. That is, for those sediment layers where Eh is controlled by bacterial regimes, redox reactions will increase certain dissolution reactions (such as Fe^{2+} formation from $Fe(OH)_3$) and precipitation reactions (e.g., FeS from Fe^{2+} and S^{-2}). Stable minerals which occur as a result of different biochemical and geochemical combinations are described in some detail in section 4.0.

Direct chemical leaching agents include materials such as fulvic acids which have demonstrated an ability to chemically desorb metals from chemical precipitates and surfaces such as clays. Contaminants which are susceptible to organic chelation will have a behaviour which is relatively independent of chemical equilibrium described under "geochemical mechanisms" since chelation will keep the contaminants in solution.

Temperature is a master variable which influences reaction rates; the rate of biological activity typically doubles over a 10°C temperature change (i.e., $Q_{10} = 2$).

1.1.1.4 Perspective on Biogeochemical Characteristics

The importance of knowledge concerning the form of tailings after mining-milling and deposition in the lake is that it fundamentally governs one's ability to make inferences about long-term processes. Especially important are the initial values of pH, Eh and ionic strength in the pore water, and the concentration of major ions and radionuclides in the pore water and on the surfaces of the particulate material. However, during the first 100 years, geochemical mechanisms and biochemical mechanisms are expected to begin to dominate. Over the long-term, these mechanisms will be the dominant control on these master pore water variables. For example, the pore water pH may be controlled by the carbonate system or by some other system (e.g., H_2S-S^{-2}), depending upon the relative rates of precipitation-dissolution reactions with particulates and input rate of materials into the tailings mass from groundwater sources.

In the tailings themselves, it is expected that geochemical mechanisms will control the long-term behaviour rather than biochemical mechanisms, if organic carbon is not present at time 100 years and if insubstantial quantities are transported into the tailings. If substantial organics are present, biological communities will develop to change the natural geochemical system to one mediated by a biochemical system. The probable controls on this balance are examined further in sections 1.3.2 and 4.0. Hydrogeological field data from Jackson and Inch (1980) provide some perspective on this for some radionuclide examples.

Prediction of geochemical mechanisms depends upon our understanding of the different mechanisms. For minerals in the sulphide group it is dependent upon an ability to predict the production of pore water concentrations of sulfide; for precipitation-dissolution reactions, it depends upon our ability to use equilibrium chemistry and the change in the surface character of the precipitates over time (e.g., amorphorus or crystalline). It depends upon whether geochemical mechanisms or transport controls pore water concentration. These are all fairly technical questions, but ones which cannot be ignored.

The power of a sub-aqueous disposal proponent's ability to predict the relative importance of different mechanisms is conditional upon his ability to quantify such mechanisms. Field studies of existing saturated tailings areas or use of simulated laboratory systems over 2-3 years are needed to confirm mechanisms and to give confidence in use of quantitative tools for predicting biochemical and geochemical

mechanisms. At present, such tools are probably capable of proper predictions within a range from 100% error to an order of magnitude or two. Fortunately, for some systems, there is a sufficient difference between mechanisms that an error of even an order of magnitude does not significantly affect such computations. The ability to predict the relative dominance of long-range mechanisms with sufficient confidence (in tailings, barriers, and in sediments) is probably one of the weakest links in the whole area of assessing the viability of deep lake disposal of U-mill tailings.

A pilot study which takes existing knowledge of equilibrium chemistry, of geochemical mechanisms, and of different biochemical mechanisms and examines the possible long-term controls for different possible combinations of tailings-sediment systems may be in order. Such a study could take available data for different systems (e.g., pH, Eh, ionic strength, levels of organics, levels of SO_4 and S^{2-}) from fresh water systems (e.g., Nriagu and Dell, 1974), salt water systems (e.g., Berner, 1975) and groundwater systems (e.g., Inch and Jackson, 1980; Cherry et al, 1980), and attempt to obtain reasonable agreement between theoretical models and field observations. Such a study would permit statements about the expected confidence that a proponent and the AECB could have in our present ability to predict the long-term behaviour of geochemical and biochemical mechanisms in tailings-sediment systems. Various example situations are given in section 4.0.

Also, a separate study could be made on natural laboratories, i.e., lakes previously impacted by tailings. Use of models derived from the pilot study described above plus field work on existing Canadian lakes previously subjected to runoff from tailings areas or subjected to direct discharge of tailings to the lakes would form a basis for actually checking these models against a "natural laboratory".

1.1.2 Physical Characteristics of Tailings

The physical characteristics of tailings will influence their initial rates of consolidation or compactness, their long-term consolidation, and the transport characteristics through them. Their physical characteristics would actually control the initial compactness and long-term consolidation, provided chemical charge considerations (e.g., repulsive forces) are not significant. The direction and rate of transport will be controlled i) by hydraulic gradients of the regional groundwater-lake-system and ii) by the relative effects of permeability and of diffusion in the tailings-sediment system. These integrated aspects are examined in section 4.0.

1.1.2.1 Initial Characteristics upon Deposition

Possible Measurements

1. particle-size distribution
2. density
3. organic-inorganic ratio
4. settleability
5. consolidation rate

Initial physical characteristics of the tailings influence the compactness (e.g., void volume) of the tailings and the interconnections of the void spaces. Measurements from which inferences can be drawn regarding this compactness include particle-size distribution, (possible void volume), specific and bulk density (a measure of their ability to compact), organic-inorganic ratios (organics will possibly prevent close packing of inorganic particles), settleability, and consolidation rate.

1.1.2.2 Porewater Transport Characteristcis

Possible Measurements

1. vertical and horizontal permeability coefficient
2. diffusivity coefficient

The vertical and horizontal permeability coefficient and the diffusivity coeffient are measures of the relative amounts of transport due to advective and diffusivity processes. The actual amount of transport is also a function of the hydraulic gradient and directional concentration gradient respectively (see section 4.0). Long-term consolidation will directly affect the permeability coefficient, but not the molecular diffusivity coefficient. The bulk diffusivity coefficient would be affected if the degree of interconnection of void spaces changes. The application of these concepts is further explored in section 1.3, 2.1.1, and 4.0).

1.1.2.3 Stability and Cohesiveness

Possible Measurements

1. stability tests in laboratory
2. cohesion tests in laboratory

These parameters describe both the "foundation" properties of the tailings and their ability to resist sudden accelerations. Such accelerations may be caused by current-driven erosion of the lake bottom or by geotechnical extrema such as earthquakes. Such information is useful if the acceleration causes substantial shifting of the tailings. They are standard tests in foundation analysis; their adaptation to an underwater environment has been made for several applications (e.g., water treatment plant diffusers). This adaptation may be sufficient for tailings disposal questions.

1.1.2.4 Solids Transport Processes

Possible Measurements

1. consolidation processes
2. solid diffusion coefficient

This section examines those processes which affect the long-term, slow movement of the particles of tailings rather than those processes which cause short-term, rapid acclerations. Consolidation processes represent the continued repacking of a tailings mass due to the effects of gravity and gradually increasing weight of deposited overlying sediments. This results in a decrease in void volume and a higher tailings density. The consolidation process may not be significant in many situations if the weight above does not increase substantially over a 10 000 year time scale.

Solid-state diffusion represents the migration of materials within a lattice or mineral due to solid molecular processes. In cement, typical migration rates of radionuclides are of the order of 10^{-10} cm^2/sec (e.g., Godbel, et al, 1980), while those in igneous rock are of the probable order of 10^{-12} cm^2/sec. For isotopes whose half life is of the order of 100 000 yr and which are contained in 1 mm particles, such migration can be the rate controlling step in migrations, of radioisotope if chemical dissolution of the particle is relatively insignificant.

1.1.2.5 Perspective on Physical Characteristics

Physical characteristics are useful descriptors of chemical properties where these characteristics are a function of particle-size distribution. They control the permeability through the tailings, but the actual advective transport is highly influenced by regional groundwater gradients and permeabilities of adjoining sediments and geological formations. Pore water diffusivity coefficients are generally independent of physical sediment characteristics except for the temperature of the tailings, but they may be affected by biological activity, e.g., bioturbation.

1.1.3 Perspective on Mine Tailings in Sediments

The assessment of whether or not geochemical or biochemical reactions are likely to occur within a tailing mass must consider i) the present oxidation state of process products and accessory minerals, ii) the redox environment into which the tailing mass is being placed and only change in that redox environment which is likely to occur in the future, iii) the presence of a nutrient and carbon source within the tailings mass which would allow the development of a microflora that could exploit energy gradients, iv) the products that would be produced by either chemical or biochemical redox reactions and their solubility, v) the effect redox reaction products may have on the effective permeability of the tailing mass or any barrier which may surround it, and vi) the chemical or biochemical change that may occur when a soluble contaminant crosses a chemical boundary such as a redox gradient as it leaves the tailing mass.

During an initial time period, these factors will essentially determine the dominance of geochemical or biochemical mechanisms and their effects upon pore water master variables such as pH, Eh, ionic strength, major elements and radioisotopes. Over the long term, these factors will continue to be dominant; however, the rate of transport of certain materials (e.g., organic carbon, nitrate, oxygen, sulfate, calcium carbonate, alkalinity) into the tailings mass may begin to become influential. Also, the rate of outward transport of pore water constituents will become influential. If the transport rate out of the tailings of all constitutents is relatively fast, those materials which are able to dissolve most quickly may control these master variables - but when they become exhausted, they will be replaced by other materials.

The final predicted form of the contaminant is essential information for estimating its subsequent bioavailability if it enters the biosphere.

When some sort of definitive statement can be made about the above points this data can then be coupled with impinging environmental variables such as local and regional hydraulic gradients and flow lines, local groundwater and surface water chemistry, and overlying lake status to predict environmental dispersion characteristics and biological effects. This is considered further in sections 4.0 and 6.0.

1.1.4 Relationship to Biogeochemical Cycle

Knowledge of the material mined, of processing data, and of the method of placement and environmental modification due to environmental change or due to engineering manipulation in the short term allows one to make estimates of

a) the characteristics of tailings initially placed in the lake, and
b) the potential of a contaminant for entry at a given rate into biogeochemical cycles.

1.2 TAILINGS BARRIER

If a contaminant is leached from the tailings, a tailings barrier, if constructed, becomes the first line of defence for controlling the extent of its entry into the lake system. The extent of transport is a function of the permeability and diffusivity coefficients of the barrier itself and of the regional groundwater field, assuming that transport of soluble forms of the contaminant is the only significant mechanism. The long-term physical and chemical properties of the barrier are described here; the actual net transport through the barrier is considered in section 4.0.

If significant movement of tailings into the liner or incursion of the liner into the sediments occur due to such an extrema an earthquake or a large lake current, then transport of particulate materials will occur. The incidence of this transport and effects upon dissolution-precipitation of contaminants between the particulate material and porewater would need to be included in an overall examination of lake disposal of U-mine tailings.

1.2.1 Physical Characteristics

1.2.1.1 Consolidation Processes

Possible Measurement

1. Long-term consolidation rates

Under an external load, various materials consolidate by repacking the individual grains and by forcing water out from the voids. Consolidation decreases porosity and permeability and, hence, transport rates through the barrier. Data used for foundation design for similar kinds of materials and for barrier materials in wastewater lagoons would permit estimation of the long-term rate of consolidation; laboratory testing may also be essential.

1.2.1.2 Stability

Possible Measurements

1. cohesiveness
2. ability to resist seismic activity

Under a lake system, a barrier will generally be under compressive load and would not be expected to have to resist tensile forces. If for example, large bottom currents are induced before lake sediments cover up the barrier, or if bottom sediments remain thin for a long period, the bottom shear stresses exerted may disturb the surface layers of the barrier. The cohesiveness of the barrier material, its thickness, and the frequency of such bottom currents are factors which may be used to assess the barrier's long-term stability.

Seismic activity may be sufficient to disturb the barrier at some point in the future. Tests carried out on barrier materials using a horizontal or vertical seismic shaker table would give relevant information. Few of such tables presently exist in Canada or the U.S. Tables capable of simulating both horizontal and vertical motions simultaneously are inadequate at present.

1.2.1.3 Porewater Transport Characteristics

Possible Measurements

1. permeability coefficient
2. diffusivity coefficient

If the tailings barrier is constructed under water, its properties can be estimated using data available for earth dams constructed underwater. Data for similar techniques of underwater construction should be used. These data should be modified to take account of long-term consolidation processes. The direction and quantity of flow may be calculated using suitable groundwater models and knowledge of adjoining strata and piezometric gradients.

1.2.2 Chemical Characteristics

1.2.2.1 Uptake Characteristics

Possible Measurements

1. chemical nature of barrier material
2. uptake mechanisms

This discussion assumes that contaminants of concern found in the barrier materials are carried there by transport from the tailings mass.

The chemical nature of the barrier materials refers to the type of surfaces which radionuclides, heavy metals or other leached materials (e.g., H^+, S^{-2}) would "see" as they are transported out of the tailings through the barrier. Clay materials have a high adsorptivity for radioisotopes. A fine-grained collophane apatite has even higher adsorption capacities. A mixture of the clay and apatite makes an excellent adsorber for radium.

In addition to adsorption, two other mechanisms which might be significant in removing migrating radionuclides and sulfide from porewater are chemisorption and chemical precipitation. If the porewater is anoxic, iron sulfides and other metal sulfide precipitates are the stable phases. A potential for chemisorption or coprecipitation of a radionuclide with these precipitates may exist depending upon the chemical concentrations of the porewater. These metals might dissolve from precipitates such as FeS formed in the barrier if no more material, e.g., sulfide, is leached or if the groundwater redox potential changes in the future. Deliberate introduction into the tailings mass of other elements capable of forming minerals with barrier material may be an advantageous engineering control measure.

Adsorption data for various clays is available in various references (e.g., Landa, 1980). More information is needed to double-check these data.

1.2.2.2 Stability

Possible Measurement

1. consolidation characteristics of clay mixtures

The compaction of clays is strongly affected by the surface charges on the clays. For example, kaolin compacts much worse than a mixture of clays which include kaolin. This is caused by the unimodal grain-sized distribution and uniformity of charges in simple kaolin. An ideally-compacting barrier should have alternating lattices of positive and negative charges. Examination of lakes sediments in nearby watersheds which export clays to their lakes provides some evidence for the effects of charges and organics on potential consolidation characteristics. Selection of clays (if they are to be used for a barrier) from the drainage basin of a lake with well-compacted clay sediments for testing of its deconsolidation-consolidation character would be in order.

1.2.3 Perspective on Tailings Barrier

The tailings barrier, if constructed, provides a first line of defence against movement (porewater transport, displacement of particulate associated contaminants) of leached contaminants from the tailings area in significant quantities. Assuming that all contaminants of concern come from the tailings mass, there are three questions of concern. First, if the tailings are leached, how fast is the material moved through the tailings barrier and how does this compare to background fluxes of the same material through the lake? Second, if the tailings particulates or solutes are moving through the barrier, how much is removed from the pore water to the barrier particles through such mechanisms as co-precipitation and adsorption? Third, if a perturbation occurs, how much is the transport of the contaminants changed, for what period of time, and what is the relative magnitude of this change compared to the long-term transport processes? The answers to these questions would permit a perspective on the quantity which may reach the lake water if no lake sediments exist.

Information concerning mechanisms and properties through a barrier were considered in this section; the actual flux of contaminants through the tailings can be calculated using the information discussed in section 2.2 and 4.0.

1.3 LAKE SEDIMENTS

Lake sediments act as either biogeochemical sinks or sources of trace elements and regulate the release of potential contaminants to the water column and to the biological community. Over the long term it is a reasonable hypothesis that transport of soluble contaminants will be the most significant form of movement of these contaminants through sediments. Periodic perturbations may also cause significant transport by physically moving sediment particles to which contaminants are attached. Advection (bulk movement of water through sediments) and diffusion (chemical migration due to molecular bombardment) will be the main transport mechanisms of soluble materials. The relationships between soluble concentrations and particulate concentrations are governed by geochemical and biochemical mechanisms. The actual flux of soluble materials out of the sediments to lake water is controlled by the relative balance of transport, biochemical and geochemical processes.

Trace elements including potential contaminants are adsorbed on, co-precipitated with, or occluded within various inorganic and organic matrices and become incorporated into sediment. Physical properties governing the sediment-water distribution of trace elements include cohesiveness, density, diffusivity, porosity, and the flow of interstitial water. Physical variables such as particle size distribution may be considered as both a physical and a geochemical variable due to correlations found between chemical properties and grain size of many sediments.

Important geochemical considerations include the organic matter content of sediments; the molecular nature of organic and inorganic sediment constituents; pH and buffering capacity; reduction-oxidation potential; and concentrations of dissolved oxygen, sulphides, hydroxides, carbonates, phosphates, and dissolved organic matter in pore water. Elemental properties such as ionization potential, electronegativity and ionic radius also affect contaminant-particulate interaction. Other processes imposed upon these controlling factors include uptake of contaminants from sediments by biota, physical disturbance of sediment by biota (bioturbation) and the interaction of lake currents (2.1.1) and sediments.

Since sediments provide the first line of defence against contamination for a tailings mass in the form of a natural or engineered barrier, and since they control the bioavailability of potential contaminants, a

more thorough inspection of controls on sediment-contaminant interactions is useful. This is done firstly by examination of transport of soluble and particulate materials and secondly by examination of geochemical and biochemical processes and their interrelationships. A few aspects of relationships between transport and these biogeochemical processes are also examined.

1.3.1 Lake Sediment Physics

In this section, the physics which affect long-term migration of tailings contaminants through lake sediments are examined. It is divided into four areas - consolidation processes, transport mechanisms, sediment stability and accumulation characteristics. Physical properties of sediments are considered as they affect sediment physics.

1.3.1.1 Consolidation Processes

Possible Measurements

1. long-term consolidation rates
2. consolidation factor in sediment transport models

Similar to the physical processes of the tailings and of the tailings barrier, lake sediments undergo long-term consolidation processes. Upon initial formation, the sediment matrix structure attains an equilibrium with the overlying water. Hence, subsequent consolidation is affected only by the weight of new material added to the sediments, assuming that bottom currents are insignificant. For many Canadian Shield lakes, consoli- dation changes the sediment porosity significantly; for example, surface sediments of Canadian Shield lakes will have a porosity of 0.9 while that of deeper sediments will be of 0.7 to 0.8. Such changes are typical of sediments whose organic content is of the order of 30-50% dry weight.

Two possible methods for determining consolidation rates are laboratory testing and use of sediment transport models. Laboratory testing would involve obtaining existing lake sediments and conducting consolidation tests upon disturbed and undisturbed sediments; judicious selection of sediments from shallow depths and imposing a load using similar sediment for the loading is perhaps the best surrogate of several different test possibilities. Since the structure of sediments is

highly dependent upon the matrices created by organics, extrapolation of any test is difficult because of the unknowns caused by chemical interactions between the organics. A second possible measurement is an inferred rate of consolidation (consolidation factor) from sediment transport models. Such models describe the accumulation of sediments due to lake sedimentation and the change in porosity over depth due to this accumulation (e.g., Imboden, 1975). These differences may be used to describe the rate of consolidation.

1.3.1.2 Transport Mechanism Through Pore Water

Possible Measurements

1. vertical advection
2. vertical diffusion

Advective flow and diffusivity will be two main processes controlling the long-term transport of soluble materials from tailings.

Advective flow is caused by long-term consolidation (squeezing, e.g., see Imboden, 1975) and groundwater flow. Knowledge of the regional groundwater field is necessary to calculate advective flow. Sediment properties such as mineralogy, grain size distribution, porosity, density and percent organic matter affect the sediment permeability. Biological processes may significantly influence advective flow within the biologically active sediment layer. Many species of oligohaetes and chironomids actively circulate water through their burrows. This process could accelerate advective transport and reduce distances over which diffusion gradients are dissipated.

Diffusion is normally molecular in sediments. Accordingly, use of the diffusion coefficient for a soluble element is a first-order approximation. Use of a heat conduction coefficient as an estimate is inappropriate because such effects as an element's ionic radius makes the numeric value of its intrinsic diffusivity coefficient much lower. The effects of chemical potential gradients of other materials also influence the value of the diffusivity coefficient. These chemical potential effects are particularly important for estimating transport and diagenetic processes over a 10 000 year timescale. Use of sediment dating together with sediment pore water models (e.g., Goldberg and Koide, 1963, Lerman and Taniguichi, 1972, Imboden, 1975, Berner, 1975, Nriagu, 1972, 1976, and DiToro and Connolly, 1980) permits estimation of both the diffusivity coefficient, the advection rate and the consolidation rate.

1.3.1.3 Stability

Possible Measurements

1. ability to resist shear forces (e.g., shear tests)
2. macroinvertebrate densities (bioperturbation effects versus depth of habitat zone)

Stability refers to the ability of the sediments to resist upset due to lake currents, seismic activity, and burrowing by macroinvertebrates (bioturbation). If significantly affected, the porosity may increase or the sediments may be mixed with underlying material such as the tailings barrier or the tailings themselves.

An ability to resist shear forces (cohesiveness) is a measure of a sediment's ability to resist erosion or seismic activity. Various static stress-strain tests are available but these are limited by their static nature and by the ability of the test to simulate the actual supporting forces in a real lake sediment. Dynamic tests under various shear force scenarios would be more realistic; however, such tests are mainly research tools - static tests do give the engineer an ability to make educated inferences concerning stability.

Macroinvertebrate densities affect stability mainly by increasing the pore water area. Their tunnels could significantly decrease the sediment stability but no literature was found which examines this question quantitatively.

1.3.1.4 Accumulation Characteristics

Possible Measurements

1. depositional or erosional nature of environment
2. ratio of sediment accumulation rate to lake sedimentation rate
3. fingerprint features of existing sediments

Within a lake, shallow nearshore waters may be zones of sediments accumulation but overall they tend to be zones of little accumulation or of erosion. The sediments of deeper waters are generally accumulation zones. Location of tailings in the deepest waters would minimize the likelihood of erosion. If the tailings fill up a large fraction of the lake, a significant redistribution of depositional-accumulation characteristics may occur.

The work of Hakanson (1980) provides some quantitative evidence for the differentation of depositonal from erosional sediment characteristics. His work is considered in Section 2.2 in more detail. Coarse parameters such as grain-size distribution can also be used. We suggest that the ratio of the accumulation rate of a local sediment area to the average lake wide accumulation rate (lake sedimentation rate) is one useful parameter for examining within-lake differences; care must be used to remove the effects of consolidation and losses from the lake through the outflow due to erosion in the discharge river area of the lake sediments.

Fingerprint features such as varved sediments and the occurrence of minerals not at equilibrium with pore water constituents are two phenomena which can be used to assess the accumulation characteristics. For example, varved sediments permit arguments concerning the lack of significant erosive processes in the past. Minerals which are consistent with pore water and diagenetic processes of other areas of the lake have probably been carried from an erosional sediment to their present location by some extreme event in the past. Some care must be used with such fingerprint features - other possible sources of a mineral such as the watershed must be considered before such inferences can be drawn.

1.3.1.5 Perspective on Lake Sediment Physics

The flux of soluble constituents through a lake's sediment is expected to be the dominant mechanism for movement of potential leachate from U-mill tailings through the sediments. Sediment physics describe this movement; biogeochemistry specifies the relationships between the concentration of a material in the soluble phase and the mass in the particulate phase.

Advection and diffusion will be the two main mechanisms of transport of soluble materials. The relative importance of advection and diffusion is dependent upon consolidation processes, the regional groundwater field, and disturbances such as bioturbation and seismic activity. Long-term diffusion may be the dominant process, depending upon physical and chemical properties of pore water. Physical movement of the particulate phase would remove contaminants on the particles from the sediment area. Whether particle movement is important in the long term depends upon the distance of movement, the direction of movement (e.g., into lake water, into the tailings mass), the biogeochemical solubilization rates, and mechanisms which become activated for removing these solubilized materials from the water.

1.3.2 Lake Sediment Biogeochemistry

In this section, those biogeochemical characteristics of sediments which are important factors regulating the sediment-water distribution of potential contaminants from U-mine tailngs in lake sediments are considered.

1.3.2.1 Geochemical Processes

Possible Measurements

1. correlations between contaminants and physical or chemical sediment variables
2. correlations between pore water elements and particulate phase parameters
3. minerals observed in sediments and potential for reacting with migrating contaminants
4. pore water concentrations of elements involved in diagenetic processes

Forstner (1977) describes a variety of geochemical metal associations in sediment:

1. Heavy metals are transported and deposited as constituents in the natural inorganic detritus, mostly in inert positions.

2. Slightly alkaline waters promote the formation and precipitation of hydroxides, carbonates, sulphides and phosphates of heavy metals.

3. Sorption and cation exchange occur on fine-grained particles with large surface areas. The general sequence of the sorptive capacity of solids is MnO_2 > humic acids > hydrous Fe oxides > clay. The sorptive capacities of different clays also vary (Buorg and Filby 1974; Farrah and Pickering 1977).

4. Dissolved organic compounds complex with dissolved metals. Less condensed humic acids (fulvic acids) are particularly important in complexing with heavy metals due to their large number of functional groups.

5. Heavy metals may co-precipitate with hydroxides and carbonates. Heavy metal carbonates having low solubilities will be completely precipitated from solution as a result of co-precipitation with calcium carbonate. Iron hydroxide concentrations can have metal enrichment factors of 30 times for Cd, 17 for Zn and 1.4 for Cu in comparison with clays in the surrounding sediments. Similarly, calcium carbonate coatings in river sediment can have enrichment factors of 10 times for Cd, 6 for Cu, 4.5 for Zn and 4 for Co when compared to geochemial background concentrations in carbonate rocks. Specific factors for radionuclide elements are only now being evaluated.

These geochemical associations of trace elements in sediments are, more or less, reversible associations. A myriad of factors influences the relative importance of these various associations, and consequently governs the sediment-water distribution of potential contaminants.

The following geochemical variables influence sediment-water distributions of contaminants:

Sediment properties
- particle size distribution
- adsorption isotherms
- mineralogy
- concentrations of potential contaminants

Pore water chemistry (vertical profiles)
- pH
- buffering capacity
- redox potential (Eh)
- concentrations of major ions
- concentrations of elements in pore water

Concentrations of many trace elements in sediment particles are correlated with the sediment particle size. For a given sedimentary component (e.g., a clay) the relative surface area and the number of potential contaminant binding sites on particle surfaces is inversely correlated with sediment particle size on a mineralogical basis. Many studies of contaminated marine and freshwater sediments have demonstrated this type of relationship between contaminant level and sediment particle size.

A particle size analysis of sediments from various depths of a lake should be carried out to assess the relative ability of the sediment to bind introduced contaminants. An understanding of the zonation of sediments throughout the lake will permit a prediction of the areas which will receive the greatest inputs of contaminants to the sediments.

A second important characteristic deserving consideration is the sediment adsorption isotherm. Adsorption involves the accumulation or concentration of substances at a surface or interface. Adsorption from a solution onto a solid occurs as a result of one or two characteristic properties of the sediment. These properties are the hydrophobic character of the solute and the affinity of the solute for sediment. Adsorption may be due to physical or to chemical interaction, depending upon solute characteristics, sediment characteristics, and environmental conditions such as pH, Eh, and temperature. Positive adsorption leads to the removal of solutes from solution to the surface of sediment particles until a dynamic equilibrium is established. At equilbrium, there is a defined distribution of solute between liquid and solid phases. This distribution is dependent upon the concentration of the solute, the concentration and nature of competing solutes, the nature of the solution, the type of sediment, etc. The adsorption isotherm is the mathematical expression of quantity of solute adsorbed per unit weight of sediment over a range of solute concentrations. More detailed descriptions of adsorption processes are provided by textbooks such as Weber (1972). Adsorption theory from any field is useful here; the main differences are the actual adsorption characteristics of the contaminants and sediments in question.

Adsorption isotherms for sediments and potential contaminants from uranuim tailings could be determined. This would provide a measure of the sediment-water distribution of potential contaminants for the pH range, the Eh range, and the different sediment types expected in lakes. Long-term studies should be made (1-3 yr.) to ascertain the long-term adsorption behaviour. The evaluation of different contaminant mixtures and concentrations expected in the long term would provide information on the sensitivity of long-term processes to significant shifts in sediment properties.

The mineralogy of inorganic sediment constituents exerts an influence on the sediment-water distribution and may warrant consideration. The adsorption isotherm of a sediment is largely dependent upon the mineral form and the presence of other constituents such as organics.

An important characteristic of sediment minerals is cation exchange capacity. Ion exchange is one of the principle types of sorptive reactions. For example, Jackson and Inch (1980) reported that three-layered expansible clays such as montmorillonite and vermiculite show very high sorptive capacities for radionuclides. Quartz is a relatively poor adsorbent of radionuclides while feldspar is a relatively good adsorbant. Furthermore, oxides, sulphides, carbonates and chlorites on the surface of granitic rock are potential radionuclide sinks.

Diagenetic formation or breakdown of sediment minerals will also influence sediment-contaminant interactions. For example, pore-water acidification will induce the breakdown of montmorillonite if the pH is sufficiently low, thereby altering the sorptive characteristics of sediments. The formation of iron and manganese phosphate minerals in aerobic sediments may lead to the co-precipitation of phosphates and other trace metals.

Sediment mineralogy may be determined using techniques such as x-ray diffraction, chemical analysis, etc.

The bulk sediment load of potential contaminants such as metals and radionuclides is an important measure to assess background concentrations prior to tailings disposal, to monitor the establishment of steady-state concentrations after disposal, and to map contaminant transport patterns throughout the drainage basin. Selective dissolution analysis of trace contaminants from sediments will provide an indication of the proportion of sediment contaminant loads which are potentially more mobilizable and, therefore, may be more readily transported or accumulated by biota. Selective dissolution analysis has been pioneered by Forstner and Muller at the Sediment Research Institute of the University of Heidelburg (see Forstner and Muller 1976). Extraction of contaminants using conditions simulating severe acidification may be appropriate for assessing mobilization from poorly- buffered Shield lake sediments.

An understanding of pore water chemistry in lake sediments is essential to the assessment of sorptive capacity and diagenetic reactions occurring under natural conditions. Samples of pore water can be taken using sediment peepers placed in situ for two weeks. (Peepers sample pore water at 2 cm intervals. They provide a powerful technique which have only recently been developed and have been used by BEAK personnel in recent research studies.)

Of prime concern, particularly in poorly buffered Shield lakes, is pH. pH changes may affect diagensis of mineral constituents, adsorption capacity, organic complexing capacity and precipitation reactions. It thereby alters the sediment-water distribution of contaminants. A decrease in pH also leads to the dissolution of metal carbonates and hydroxides. At low pH's, protons compete for adsorption sites on sediment particles. The clay kaolinite removes all Cu (II) from solution at pH >6 while at pH <3.5, no Cu uptake by the clay occurs (Payne and Pickering 1975). Similar observations were made by Farrah and Pickering (1977) and Frost and Griffin (1977) who studied the removal of various metals from solution by clays. Shifts in pH may also alter the solubility of organic matter and associated contaminants.

Since pH may vary seasonally and may be altered by the production of H_2SO_4 from the oxidation of pyrites introduced in mine tailings, an understanding of present pH conditions in the sediment profile and of mechanisms controlling pH in sediment pore water is essential.

The buffering capacity of pore waters will determine their ability to resist a shift in pH. As pH changes can greatly influence the sediment-water distribution of contaminants, a knowledge of the buffering capacity of the lake sediments is fundamental to understanding the implications of deep lake disposal. Alkalinity of waters usually refers to the quantity and kinds of compounds present which collectively shift the pH to the alkaline side of neutrality. Alkalinity is usually imparted by the presence of bicarbonates, carbonates and hydroxides and less frequently by borate, silicate and phosphates. The CO_2-HCO_3-CO_3 system is usually the major buffering system in fresh water; lakes on the Canadian Shield where most deep lake disposal would be potentially carried out are poorly buffered due to the lack of carbonate minerals in their watershed.

In sediment pore waters of low alkalinity lakes, sedimentary materials such as alumino-silicate minerals (clays) can play an important role in buffering. However, competition for binding sites on sediments by protons will cause dissociation of metal contaminants and potential release to the water column. Weak organic acids such as tannic, humic and fulvic acids will build up in sediments due to the decomposition of organic matter. These acids will buffer the effects of added protons to some degree by forming organic complexes; however, the association of cations and radionuclides will be broken leading to

mobilization. H^+ movement from between layers of clays provides some buffering capacity; at low pH, these clays break apart liberating H^+ and other ions, reducing the potential for clays to act significantly as a portion of the buffering system.

To assess the susceptibility of poorly buffered lakes, to pH shifts due to acidic precipitation or to acid generation from pyritic mine tailings, the measurement of bicarbonate and total alkalinity and the mathematical description of potential pH shifts should be carried out on low alkalinity lakes where tailings disposal is proposed. Methods for this are assessed in section 5.2.

Pore water concentrations of major ions such as Fe, Mn, CO_3^{2-}, HCO_3^-, SO_4^{2-}, S^{2-}, PO_4^{3-} are important because they influence biochemical reactions (discussed in 1.3.2.2) and because they may result in the geochemical precipitation of cationic contaminants. Metal hydroxides, carbonates, sulphides and phosphates may form and precipitate, particularly under slightly alkaline conditions. Metals may also be co-precipitated with hydroxides and carbonates of major cations. Metal carbonates with low solubilities precipitate from solution due to co-precipitation with calcium carbonate (this may be biochemically induced). Iron and manganese hydroxides precipitate under oxidizing conditions, leading to the co-precipitation of other metals. The stability of these major ions is dependent upon redox conditions, which in turn are strongly controlled by biochemical mechanisms.

Pore water concentrations of potential contaminants should be monitored, because it is this fraction of the total sediment contaminant load which is most readily mobilizable and available to the biological commmunity. These concentrations can be measured using leaching and bioavailability tests or using pore water peepers. The "Standard elutriate test" was devised by the U.S. Environmental Protection Agency for the extraction of the leachable contaminant fraction from sediments. Sediment and water are combined in a 1:4 ratio, vigourously shaken for 30 minutes, allowed to settle for 1 hour, and filtered or centrifuged. Concentrations of contaminants are determined on the elutriate. Physico-chemical conditions in the water (e.g., pH, Eh) may be controlled to best simulate conditions in the overlying water (see Lee et al 1978). Procedures for elutriate testing are detailed elsewhere (Keely and Engler 1974; Army Engineer Waterways

Experiment Station 1976; USEPA/Corps of Engineers 1977). While elutriate testing is designed primarily for monitoring sediment quality for dredging purposes, its use or that of chemical extraction procedures (e.g., Argemian and Chau, 1977; Kilborn, 1980) provides an estimate of the mobility of various contaminants in lake sediments and their interaction with mobile contaminants from tailings.

For elutriate testing, we would suggest application of tailings leached contaminants to water overlying intact sediments in laboratory lysimeters. Techniques similar to studies at the C.C.I.W., W.T.C. (Environment Canada, Burlington; unpublished results) may be employed. Different redox environments and different source strengths should be applied under saturated conditions, rather than the percolation approach used by C.C.I.W.. Elutriate tests may then be performed upon the sediments; further information would be obtained by use of pore water peepers in the laboratory columns.

1.3.2.2 Biochemical Processes

Possible Measurements

1. percent organics
2. redox potential, $[O_2]$, $[S^{2-}]$
3. dissolved organics in pore water
4. source of sediment organics
5. stability of sediment organics (biodecomposability, sediment oxygen demand)

Biochemical systems in lake sediments directly and indirectly control the sediment-water distribution of metal contaminants. The different aspects are examined here in depth; the overall picture is examined in a simpler sense in section 4.0.

As described in the previous section, the organic content of sediment provides a good indication of its sorptive capacity for introduced contaminants. Organic compounds generally have a very high cation exchange capacity owing to the presence of ionizable functional groups (e.g., COOH). Thus, sediment metal levels tend to be directly related to organic or carbon content (e.g., Kennedy et al, 1971; Cline et al, 1973; Hutchinson and Fitchko, 1974; Townsend et al, 1974; Ramamoorthy and Rust, 1976, 1977; Jackson, 1977; Ongley, 1978; Ouellet, 1979).

More fundamental is the role that organic matter plays in the biological systems of sediments, particularly surficial sediments. The nature of biological systems controls the Eh of surficial sediments while Eh controls most of the chemical reactions involving uptake and release. The combined effects of biological and chemical systems influences the geochemical processes (e.g., vivianite formation from Fe^{2+} and S^{2-}) of deeper sediments. Organic nutrients are present. Typically in a lake sediment, one finds a gradation from relatively easily degradable organics in surficial sediments to relatively complex stable organics in deeper sediments. In the absence of oxygen, correlations are occasionally found between biodecomposability of organics and redox potential. If one compares lakes, those with relatively more refractory organics have a higher Eh than those with more decomposable organics. A measure of these differences is the biological sediment oxygen demand.

Organic matter is also important because it provides a measure of the depositional-erosional nature of the sediment. Areas which show the highest organic content will be exposed to minimal physical disturbance and may, therefore, be the most suitable areas for tailings disposal.

Organic content is usually determined by loss-on-ignition, which removes organics through oxidation.

The reduction-oxidation (redox; Eh) profile in unconsolidated sediments is one of the most important measures of sediment quality since redox potential controls the behaviour of contaminants in sediments. Perceived Eh-contaminant interactions in sediments and tailings are illustrated in Table 4.0.1 (Section 4.0). Redox potential is proportional to the equivalent free energy change per mole of electrons associated with a given reduction (Stumm 1966). Using concepts of electron and proton activity, a discussion of the theory of the redox potential is given by Wetzel (1975) and Jackson and Inch (1980).

As dissolved oxygen concentrations approach zero in pore water (and possibly also in the hypolimnion), Eh declines precipitously. In sediments, high oxygen demands are caused by high organic matter content and by high rates of biological oxidation. The Eh will reach approximately 0 Mv or negative values within a few millimetres of the interface. Ferrous iron in the sediments is the most important reducing material in most fresh water sediments. Reduced manganese also contributes significantly to a low Eh in reducing sediments. In narrow zones, ammonia production may be significant. Sulfate is generally important only in sea water sediments. Elevated concentrations of humic substances also contribute chemically to the low Eh's, due to their reducing properties.

Khalid et al (1977) reviewed the literature concerning metal transformations as affected by oxidation-reduction. Oxygen deficiency leads first to the dissolution of hydrated manganese oxide and then to the dissolution of hydrated iron oxide. Tessenow and Baynes (1975) and Michard (1975) concluded that remobilization of Fe and Mn in anoxic sediments followed by upward diffusion may account for accumulations of Fe and Mn oxides at the oxidizing-reducing interface. Since Fe and Mn are soluble in their divalent states, other metals that were co-precipitated with Fe and Mn oxides tend to be remobilized. Reinhard and Forstner (1975) demonstrated that Cu, Zn and Cd may be released by anoxic sediments into over-lying water, although it has been suggested that heavy metal release under anoxic conditions may be attributable more to bacterial activity rather than to ion migration (Cline and Upchurch 1975).

Opposing the tendency of heavy metals to be released by oxides of Mn and Fe under reducing conditions is the process of metal reaction with sulphides to form stable compounds in sediments. Many heavy metals such as Hg, Zn and Cu are firmly bonded and precipitated as sulphides and may, therefore, accumulate in sulphide-rich sediments where organic decomposition has led to reducing conditions (Allin 1978). Subsequent oxygenation of the sediments or the addition of oxidants such as ferric iron may cause metal release into the aqueous phase (Bolshakov 1969). Jackson (1978) proposed that sewage be introduced with metal-contaminated effluents in tailings ponds at Flin Flon, Manitoba, since the subsequent production of sulphide would lead to entrapment of metals in the sediments. Such sulphide would probably come from proteins of organics rather than from sulfate ions in soft waters.

Redox potential is measured using a calomel (Hg_2CL_2) - platinum electrode with an appropriate meter for measuring the oxidizing or reducing intensity. Redox and dissolved oxygen profiles should be measured in sediment cores from all sediment types and in all major depth zones of the lake proposed as a tailings disposal site. These measurements should be taken at least during spring or fall turnover and in late summer during thermal stratification.

Dissolved organic compounds in pore waters will influence the sediment-water distribution of contaminants introduced by mine tailings. While the measurement of dissolved organic substances in pore waters may provide some useful information on biogeochemical behaviour of

contaminants, quantitative knowledge on the nature of metal-organic interactions is generally lacking. A major problem in the study of these interactions is the fact that, of the organic species involved, humic acids, fulvic acids and humins are the most important and form part of an extremely heterogeneous polymer system (Jonasson, 1977). However, their chemical properties are inadequately understood. Differences in elemental composition, acidity, degree of polymerization and molecular weight affect the nature of organic-metal associations.

Complexing agents such as humic acids, amino acids, etc. compete with sediment inorganic particulates for dissolved metals. The presence of complexing agents can result in the mobilization of major cations and some heavy metals from particulates into the dissolved phase (Chau and Shiomi, 1972; Barica et al, 1973; Lahar and Hochberg, 1975; Inoue and Suemitsu, 1979). Complexing agents mediate the sorption of heavy metals by sediments. The existence of ternary compounds of "surface-metal-ligand" and "surface-ligand-metal" has been demonstrated on several systems (Bourg, 1979). Depending upon the number and type of ligands involved, complexing agents may bind both with metals and with solid surfaces (Elliott and Huang, 1980).

At present, relationships of contaminants with the myriad of dissolved organics occurring in pore waters are poorly understood. However, as metal-organic associations are known to be of major importance, measurements of total organic carbon in sediment pore waters should be carried out to provide an indication of the potential for interaction between dissolved organics and contaminants introduced by uranium mine tailings. Selective dissolution analysis using methods outlined by Forstner and Muller (1976) might also be conducted to determine the dissolved organic-metal fractions; alternatively, lysimeter testing coupled with use of high pressure liquid chromatography-atomic adsorption spectroscopy might be used.

The origins of sediment organics, whether they be autochthonous (formed within the lake) or allochthonous (formed external to the lake), will also influence contaminant bioavailability and transport in sediments of lakes considered for disposal. Allochthonous organic matter in lake sediments is generally composed of refractory compounds (i.e., compounds that are chemically resistant, of low solubility, and resistant to rapid microbial degradation). In many lakes, particularly in oligotrophic lakes typical of the Shield, allochthonous organic matter may be the major source of dissolved and particulate organic carbon. Since organic matter plays such an important part in trace

contaminant biogeochemistry, priority should be given to the maintenance of a natural organic input into a disposal lake. Consequently, it may be necessary to restrict or to prohibit the harvesting of forest resources in the drainage basin of a disposal lake.

The importance of allochthonous and autochthonous organic matter in a lake may be determined by measuring C:N ratios. Dissolved organic matter of terrestrial origin generally shows a C:N ratio of about 50:1 as opposed to a ratio of approximately 12:1 from algae and plants (Wetzel, 1975). $^{13}C/^{12}C$ ratios can also be used to discern their origins since different 3-C and 4-C molecules (e.g., corn versus tree leaves) have different ratios.

The stability of organic compounds will determine the stability of organic-contaminant associations. Thus a relatively high percentage of stable or refractory organic compounds in sediments will tend to oppose the mobilization and transport of contaminants to the hydrosphere. In general, dissolved organics are metabolized rapidly (days) by microflora in comparison to slower decomposition for particulate organics (leaves in weeks, conifer needles and woody material in years). Labile components of dissolved organic matter are processed rapidly while more refractory components have much longer residence times in lakes.

The stability of organics in sediments may be measured indirectly by determining sediment oxygen demand or by measuring the dissolved oxygen profile in sediment pore waters. The latter is difficult experimentally. C:N ratios may also be determined to indicate the stability of sediment organics, with ratios generally being inversely related to stability. Carbon dating (C^{14}) and pollen profiles provide information for dating the sediments. Pollen profiles are typically associated with watershed disturbance due to the development of agriculture (e.g., the settlement of Southern Ontario in 1850 gives evidence for dating the sediments of Lake Erie). Humic substances may be separated into base-soluble humic acid which precipitates upon acidification, fulvic acid which is soluble in both acid and base, and humin which is insoluble in both dilute base and acid. Since humic substances are refractory, breakdown in sediments may be inhibited further by low temperature (in stratified lakes) and anoxia. Some understanding of the stability of organic matter present in sediments is required for assessment of the biogeochemical implications of the deep lake disposal.

1.3.2.3 Biological Communities

Possible Measurements

1. major bacterial communities and numbers
2. macroinvertebrate characteristics and densities
3. measures of bulk community properties

The interaction of biological communities with sediment regulates the sediment-water distribution of contaminants and, therefore, should be considered in the environmental assessment of deep lake disposal. Sediment-biota interactions are extremely complex. An understanding of various biological components in proposed disposal lakes is necessary in order to provide background data for the assessment of what biological changes may occur due to tailings disposal. Unfortunately, present day information is currently insufficient to permit a quantitative prediction of the biogeochemical effects of specific biota on sediment-contaminant associations. An examination of the biochemical controls on redox potential (section 4.0) and of chemical properties of sediments (section 1.1.1) are two methods for clarifying this confusion.

Microbes

Microbial populations increase greatly in numbers in the transition zone from the overlying water to the surficial sediments. Numbers of bacteria increase three to five orders of magnitude from water to surface sediments (by weight) and decrease rapidly within sediments at greater depths (Wetzel, 1975). The bacterial biomass in profundal sediments generally correlates positively with increasing lake productivity. Bacterial populations in low productivity lakes proposed as disposal sites will be fairly low. Bacteria contribute to trace metal mobilization from sediments. Kusnetsov (1975) demonstrated that microbes are partially responsible for the decomposition of higher molecular weight organic acids, which might be associated with trace metal contaminants, to smaller, more soluble forms. Ramamoorthy et al (1977) found that bacteria compete effectively with sediment in accumulating mercuric ions from water; this accumulation has a significant effect on the mobilization of mercury from sediment into food chains.

Bacteria also methylate Hg, Pb, As, Sn, and Se to organic forms which are more toxic to higher organisms. Many of the factors contributing to methylation processes are not fully understood. Formation of radionuclide-organics is also important. The equivalent effects are due to internal irradiation rather than to chemical toxicity. Whether or not highly mobile forms of organic radionuclides are formed by microbes is not apparent from the literature.

Algae may influence the mobilization of metals from sediments to water where sediments are within the euphotic zone of the lake. For example, Ankistrodesmus and Anabaena accumulated heavy metals from sediments in experiments carried out by Laube et al (1979). Giesy and Paine (1977) found that algae may participate in the mobilization of Am-241 from sediment.

Rooted aquatic macrophytes in the littoral zone may mediate in the transport of trace metals from water to sediment. Mortimer and Kudo (1974) reported twice as much Hg in sediments in which Elodea had grown in comparison with plantless control sediments. The same workers (1977) found no relationship between Hg in sediment and uptake by macrophytes and found no indication that roots mediate in Hg release from sediments. Mortimer (1977) also showed that aquatic plants from the Ottawa River acumulate Hg from water while discriminating against uptake from sediments.

Rooted plants may also mediate in the flux of trace metals from sediment. Welsh and Denny (1976) proposed that Pb is absorbed from sediments through the roots of aquatic plants and is subsequently released into the water from decaying leaves. Mayes et al (1977) demonstrated in field experiments that concentrations of Pb and Cd in Elodea canadensis grown in the same water but in sediments with different levels of the two metals differed significantly, indicating that there was removal of Cd and Pb from the substrate. Chittenden (1976) observed lower levels of Mn and Fe in weed bed sediments than in deeper areas; this suggested biological mediation in removal from sediment. In the Moira River, a tributary of Lake Ontario, concentrations of Ni, Co, and Cu in bottom sediment were correlated with concentrations in Myriophyllum verticillatum and Elodea canadensis, suggesting biological uptake from the substrate (Mudroch and Capobianco, 1979).

The influence of aquatic macrophytes on metals in sediments is dependent upon physicochemical conditions. Patrick (1977) demonstrated that uptake from sediments by marsh plants was more related to sediment conditions for Cd than for other metals, and that the rate of uptake increased with Eh. Zinc uptake was also directly related to Eh. Mercury accumulation from sediment increased with increasing pH and with declining salinity. Uptake of Cu from sediment was little effected by Eh or pH. The role of Eh may be influenced by the availability of oxygenated sediment for root development. Some macrophytes (e.g., Nuphar sp.) may contribute to gaseous exhalation of radon through the presence of elaborate gas circulation systems.

Aquatic macrophytes can also influence sediment chemistry by altering pH, Eh, sulphide levels and sediment organic levels and composition, not only through decomposition after death, but also during active growing stages. For instance, Tessenow and Baynes (1975) demonstrated that an oxidized layer in surficial sediments was maintained by oxygen transport through the tissues of the hydrophyte, Isoetes lacustris, resulting in accumulations of Fe and Mn oxides at the oxidation-reduction interface.

Benthic Invertebrates

Through the process of bioturbation, sediment-dwelling benthic organisms can influence metal distribution and physicochemistry in the sediment. Tubificids have been considered the most important mixers of surface sediment in freshwaters (Davis 1974). Prosi (1979) observed the effects of tubificids on sediment metal distribution in aquaria. Worms caused an enrichment of metals in surface sediment due to the ingestion of deeper contaminanted sediment and the egestion of contaminated fecal matter near the sediment-water interface. The metals in this fecal matter were then gradually released to the overlying water. Boddington et al (1979) concluded that benthic macrofauna may be entirely responsible for the clearance of Hg from sediments in flowing waters. They found that the clearance rate of inorganic Hg and methylmercury from sediments was linearly related to worm density in Ottawa River sediments. The most significant loss was from the surface centimetre of sediment. Krezoski et al (1978) investigated radioactivity profiles in Lake Huron sediment and demonstrated an agreement between the depth of the surface layer of homogeneous radioactivity and the verticle distribution of benthic macroinvertebrates (primarily Oligochaeta and Amphipoda). The authors considered oligochaete defecation rates, amphipod burrowing depths and benthic densities and concluded that bioturbation of sediments by zoobenthos is important in the mixing of profundal sediments in the Great Lakes.

Fish

Fish may also influence the behaviour of trace metals in sediments. Kudo and Mortimer (1979) found that the amount of Hg accumulated by fish in aquaria is nine times greater than the amount accumulated from water in which Hg had come from bed sediments. Koyanagi et al (1978)

also demonstrated direct biomobilization of trace metals from sediment by fish. While such studies are primarily concerned with the significance of bioaccumulation to the biological community, they also indicate that activities of fish exert an influence on metals in sediments.

Due to the complexity of the biota of sediments, limnologists often make chemical measurements of biological properties from which inferences are made of the gross character of these properties. Such measurements include sediment oxygen demand, (the instantaneous flux of oxygen into sediments), the maximum uptake rate of oxygen (sediments mixed in an aerobic respirometer), the nitrification capacity (the maximum rate of nitrification, an inference concerning the viability of nitrifiers), the sulphide reduction capacity (the maximum rate of sulphide production), and the biodecomposability (the total oxygen demand of the sediments under aerobic conditions) These and other methods provide inferences about total community activity (utilization rate of energy from chemical bonds by bacteria) or the activity of specific groups of organisms (A. Klapwijk, Wageningen Univ., Holland, personal communication, 1980).

Such measurements permit estimation of the long-term characteristics of biological organisms and their associated effects upon sediment pore water properties such as pH and Eh. The actual adaptation of these techniques to assessment of the long-term biochemical mechanisms in sediments is needed but has not been thoroughly worked out. Use of these techniques in conjunction with assessment of geochemical mechanisms such as diagnenesis is one possible methodology for prediction of long-term characteristics of biochemical and geochemical processes.

Perspective on Lake Sediment Biogeochemistry

As a potential sink for many environmental contaminants, the role of sediments in governing the fate of contaminants introduced by tailings is an important consideration. These roles have been subdivided into geochemical mechanisms, biochemical mechanisms, and influences of the biological community. These mechanisms govern uptake-release reactions; transport will move those constituents which are soluble.

The dominant geochemical process over the long-term will be diagenesis. Influential geochemical variables at any one point in time include the sediment characteristics of particle size, adsorption isotherms, miner-

alogy and concentrations of potential contaminants. Sediment particle size is usually inversely related to sediment contaminant concentration and, therefore, fine sediments will provide a better contaminant sink than coarse sediments. Adsorption isotherms describe the affinity of a contaminant for a sediment by quantifying the sediment-water distribution of a contaminant over the range of dissolved contaminant concentrations tested. Adsorption isotherms of sediments in a proposed disposal lake will be necessary in any attempted quantification of biogeochemical behaviour. The mineral nature of sediments should be determined since sorption-desorption characteristics are a function of the mineral nature of inorganic sediment constituents. The bulk sediment load of potential contaminants is a final fundamental variable which should be monitored to measure steady state concentrations after disposal and to map contaminant transport patterns throughout the drainage system. Bulk sediment load should be accompanied by selective dissolution analysis to provide an indication of the potential for mobilization of contaminants from sediments.

Among geochemical characteristics of pore water, pH, Eh, and buffering capacity are perhaps the most important at any one point in time. An understanding of long-term processes is necessary to predict their long-term behaviour. Hydrogen ion activity influences the dissolution of other cations by competing for adsorption sites and affecting precipitation-dissolution reactions. An ability to resist pH changes, particularly in Shield lakes where acid precipitation is responsible for declining pH in many lakes, is of utmost concern. Acid may also be generated by reactions of pyrite within the tailings.

The carbonate-bicarbonate-carbon dioxide system usually accounts for most of the buffering capacity of lakes; however, Shield lakes are generally low in carbonates and bicarbonates owing to the lack of carbonate minerals in their watersheds. An understanding of their buffering system and the contribution of carbonates, silicates, borates, phosphates, and organic acids to total alkalinity is necessary for assessing the long-term effects of acidification. Pore water concentrations of major ions, including Fe and Mn (reduced and oxidized), CO_3^{2-}, HCO_3, SO_4^{2-}, S^{2-}, and PO_4^{3-}, should be determined since they are involved in precipitation-dissolution reacions with many trace metals. Levels of potential contaminants in pore waters should be measured since this fraction is most available for transport and bioaccumulation. Peepers are useful devices for collecting pore water to make these measurements.

Important biochemical variables are generally related to the sediment organic fraction and include percent organics, redox potential and dissolved oxygen, dissolved organics in pore water, source of sediment organics and the stability of sediment organics. Organics generally exhibit a strong affinity for trace metals; thus, particulate organics tend to demobilize metals. Metal levels are directly related to sediment organic content in many lakes. The decomposition of organic matter leads to oxygen depletion in the sediments and, consequently, to a decline in redox potential or Eh. Redox governs the solubility of Fe and Mn oxides and thereby influences the solubility of other metals which are precipitated or dissolved with Fe and Mn. Reducing conditions also result in the reduction of SO_4^{2-} to S^{2-}. Sulphides form stable, highly insoluble compounds with most metals. The importance of these systems depend upon their relative concentrations. For example, if equal concentrations of nitrate and sulfate together with organic character and bacteria are present in sediment, nitrate can be reduced to nitrate while sulfate remains essentially unaffected. This is further addressed in 4.0.

Dissolved organic concentrations in pore water mediate both the sorption and desorption of metals. They may provide a significant fraction of the buffering capacity of pore waters in the sediments of soft water lakes. An understanding of the sources of sediment organics is important. Where most organics have a terrestrial origin, (e.g., mountain and Shield lakes), it may be necessary to protect terrestrial watersheds by restricting the harvesting of forest resources or protecting against long-term soil degradation from acid rain. The stability of sediment organics (examined by source and molecular structure) gives information concerning the stability of sediment-metal associations.

Biological communities regulate the biogeochemical behaviour of contaminants for surficial sediments and influence contaminant behaviour in deeper sediments. The complexity of these relationships, make it difficult to quantify these sediment-biota interactions. Microbes are responsible for the breakdown of sediment organics and compete with sediments for contaminants. Algae are known to participate in contaminant mobilization from sediments. Rooted plants may augment transport from sediment to water and from water to sediment. Benthic invertebrates and fish may influence sediment metals by mixing the sediments. Ingestion and egestion of sediments by bottom-dwelling animals aids transport of contaminants from sediment to water. Under the assumption that the sediment biota has specifically adapted to the availability of energy for growth (in the form of chemical bond energy), measurement of

the organic concentration and the bioavailability of these organics (a biodecomposability measurement under aerobic and anaerobic conditions; Lai, personal communication, 1981) is perhaps the best surrogate presently available for measuring the activity of the biological communities.

1.3.3 Relative Importance of Transport and Biogeochemical Mechanisms

Possible Measurement

1. dimensionless numbers of biogeochemical and transport variables

The essential question concerning the relative importance of transport and biogeochemical mechanisms on controlling migration of radionuclides and other heavy metals and anions such as OH^-, S^{2-}, NH_3 through sediments is: "what is the rate limiting step?" In terms of impact on a lake water, the management variable of concern is the flux (units, gm/m^2-yr.) out of the lake sediment relative to return flux to the sediments from the lake water and relative to the outflow rate from the lake through the hydrological discharge. The flux of soluble materials through water is equal to a concentration term times a transport rate (diffusion or advection). If transport is sufficiently slow, diagenetic and other geochemical processes will control the concentration, but the rate limiting step is transport. If transport is sufficiently fast, uptake-release will not be able to keep up with its ability to remove contaminants - then geochemical processes are the rate-limiting step. For further amplification, judicious conversations with chemical engineers are suggested.

It is useful to quantify a system so as to determine the rate-limiting step. This may be accomplished using several methods. Use of leaching experiments (lysimeters) and correlation of results with flow rate are one method. Unfortunately, leaching experiments make confidence in long-term extrapolation difficult due to a lack of understanding of the fundamental mechanisms. Use of retardation coefficients has some potential; unfortunately, it does not take mechanisms into account and hence, makes extrapolation with confidence difficult due to the potential for future shifts in the relative importance of different mechanisms (e.g., adsorption versus diagenesis).

Formulation of dimensionless numbers which relate biogeochemical variables to transport variables is a third approach. Such formulations are in a stage of infancy but are being developed. Exploratory ideas have been initiated by BEAK for radionuclides. The approach is to formulate mathematical models and to develop engineering parameters from these models. Consider the following example. Steady-state transport of a radionuclide through a clay bed, affected only by advection, adsorption and radioactive decay may be described by

$$0 = v \frac{dc}{dx} - k\,(1 + \rho K_D/\Sigma)\,C$$

Here v is the velocity of flow, C is the concentration with distance x, k is the radioactive decay coefficient, ρ is the bulk density, Σ is the porosity and K_D is the adsorption coefficient. The relevant dimensionless number is $\frac{k\,(1 + \rho K \Sigma)}{V/L}$

where L is the bulk length scale. This is a ratio of chemical reaction to transport. Quantitative analysis permits analyses for the rate-limiting step. The slowest rate is then the one of interest for a 10 000 year time frame.

Further development of such formulations requires analyses of the interactions of several mechanisms controlling pore water distributions with transport.

One caution is in order. The analysis suggested above is based upon pore-water characteristics and its reactions with particulate phases. If some phenomena (e.g., acidic water) act to remove all of one constituent from the sediments, then a new chemical system will control such relationships. For such phenomena, long-term estimation of expected behaviour will require formulation of the models with respect to total sediment mass (e.g., of mobilizable iron) rather than one with respect to pore water concentrations. Formulations with respect to pore water concentrations may assume that the particulate material is an infinite source or sink for reactions involving precipitation-dissolution.

2.0 LAKE WATER BODY

The long-term questions of concern with respect to the lake water body are the following: (i) What will be the increase over natural background concentrations of radionuclides and other contaminants in lake water due to potential leaching from the tailings mass? (ii) What will be the biological implications of this additional leachate? The former is the question of concern in this project.

To assess long-term natural background concentrations, long-term changes in input rates to the lake must be determined. These are a function of watershed and atmospheric characteristics. These questions are addressed in sections 4.0, 5.0 and 6.0.

To assess changes in the lake water body due to leachate transport from the sediment-tailings area, the balance between incremental transport from the sediments, rate of transport within the water body and possible return mechanisms must be examined. This report structures this examination by first considering natural properties of lake physics and then considering the natural properties of the lake water chemistry and lake water particulates (biological and inorganic particulates) and their interactions. This structure is followed in order to ask two sequential questions: (i) If leachate enters the lake from the sediment-tailings system, will it reach surface waters? (ii) If the leachate is distributed throughout the lake, are there natural compensating mechanisms which might return the materials to the lake sediment tailings system and over what time frame?

Lastly, the net transport rate from the sediments to the lake and other sediment-water interactions are examined. This permits examination of a third sequential question: If leachate that enters the lake is returned to the sediments by natural mechanisms and if sediment transport is disturbed by lake influences (changes in circulation patterns, water chemistry or water biology), what are the factors which influence the net transport rate from the sediments to the lake? A fourth sequential question: what is the actual net transport rate? is examined in section 4.0 because of the need to include the effects of hydrogeology upon regional transport regimes.

2.1 LAKE WATER RELATIONSHIPS

2.1.1 Water Body Physics

Key questions which are relevant to this section are the following: (i) How frequently and how completely will surface lake waters be mixed with deeper waters in order to bring any potential contaminants leached from sediment areas into surface water areas? (ii) What horizontal circulations exist which may affect transport between lakes and embayments? (iii) What physical properties of water and of external forces control these circulation patterns?

Wind energy and inflowing water from rivers are the two main forcing functions driving vertical and horizontal circulation. Other forces (e.g., coriolis force due to the earth's rotation) are important in large lakes, while motorboats are important in very small lakes. Internal density differences caused by solar heating of surface waters, vertical differences caused by differential inputs of salts and total dissolved solids) and basin drag due to basin geometry resist mixing of different water masses. Circulation is in the form of currents (advection) and diffusion (eddy turbulence).

To differentiate physical transport from atmospheric effects is difficult because wind energy is a main driving force for circulation and solar induced heating of the lake is the main force resisting vertical movement in most lakes. Also after ice out, solar-induced heating removes resistance to vertical motion by heating 0°C water to 4°C, the same temperature as the bottom water temperature in many Canadian lakes. However, it is essential to make such a differentiation in order to be able to assess effects of salt-induced density characteristics (e.g., meromixis). Accordingly, this section examines the properties of lake transport. In section 6.1, atmospheric properties including winds and solar energy are examined. The effects of solar radiation and wind energy upon lake circulation are then examined in section 6.3.

2.1.1.1 Vertical and Horizontal Lake Characteristics

Possible Measurements

1. horizontal versus vertical character
2. direction of chemical gradients

A dominant characteristic of many lakes is that their main density variation is vertical on an annual basis; hence, their main resistance to circulation is vertical. This is caused by the heating and cooling of surface waters by solar radiation and air temperatures. The heating-cooling cycle induces large changes in the density of surface waters over the course of the year compared to the nearly constant density of bottom waters. Hence, one may say that lake temperatures determine the density structure of most lakes. Due to the nearly equal density of water horizontally, resistance to wind-induced horizontal currents is small relative to the vertical density differences. Accordingly, horizontal circulation is easy and rapid. That is, over the course of a year, vertical temperature differences are more significant in affecting physical properties of lakes than are smaller horizontal temperature differences.

A measure of the thermal characteristics then is the slope and stability of isotherms across the lake. If they are horizontal for most of the year, then vertical, thermally-induced characteristrics dominate.

A complication is caused by the possibility that total dissolved solids (TDS) variations may be significant horizontally or vertically. These variations result from lake interactions with the drainage basin and hydrogeology.. For such possibilities, the density structure of the lake may be influenced by or determined by the TDS variations, in addition to thermal effects. For example, TDS variations of 70 mg/L cause approximately the same variations in density as temperature differences of 0.7°C (in the 10-12°C range). To assess the contribution of TDS upon density, the vertical and horizontal isoclines may be examined. Quantitative comparison between density differences induced by TDS differences and by heat extrema permit determination of the relative effect of thermal and TDS regimes upon gross density characteristics.

2.1.1.2 Vertical Circulation Characteristics

Possible Measurements

1. frequency of vertical overturn
2. maximum depth of vertical overturn
3. rate of thermocline erosion during summer stratification
4. stability of stratification (e.g., Richardson number)

Vertical transport in a lake may occur by vertical current-like motions and by diffusivity processes. Their relative importance depends upon season and lake depth. A dominant physical characteristic of lakes is the control that the annual heating-cooling cycle has on vertical circulation. In the springtime, wind-driven surface currents often have sufficient energy to cause mixing top to bottom because the lake is isothermal (uniform temperature). That is, its uniform vertical density presents minimal resistance to wind-induced currents. As springtime heating continues, surface water temperatures become warmer than bottom water temperatures, causing the density of the surface waters to be less than that of bottom waters. The density differences cause vertical buoyancy forces which are capable of resisting vertical mixing, depending upon the energy of mixing. At some point in the spring, vertical density differences between heavier and lighter waters are sufficient to resist full vertical mixing and a stratification occurs. During stratified conditions, lighter, warmer surface waters (called the epilimnion) overlie denser, colder bottom waters (called the hypolimnion). The horizontal plane dividing these two zones is called the thermocline; the zone surrounding the thermocline is called the metalimnion. The bottom waters typically have temperatures of about 4°C, (the temperature of maximum density),while surface temperatures typically vary from 8°C to 24°C.

Maximum surface temperatures occur in mid summer. As the surface waters cool in late summer and fall, the density difference between surface and bottom water diminishes. The resistance to vertical mixing decreases such that surface currents induce mixing to deeper lake depths. This causes the epilimnion to deepen. This process is often called "thermocline erosion", "epilimnetic entrainment of hypolimnetic waters" or "thermocline deepening". Over a short period of time (one to several days) in the late summer or early fall, wind-induced surface

currents are large enough to cause complete vertical mixing. This event is called "fall overturn". During subsequent periods, complete mixing will continue until ice forms. When ice forms, an "inverse stratification" occurs in which surface waters are lighter than bottom waters because water density decreases as the temperature decreases below 4°C (this inverse stratification is weaker than summer stratification in terms of density differences with depth). Due to ice formation and isolation of lake water from wind effects, surface currents are quite small. "Inverse stratification", in combination with isolation of the lake water from wind effects, prevents vertical mixing. Upon ice thawing in the spring, the lake is again able to mix completely, if surface currents are sufficiently strong.

The mixing cycle described above is a description of "dimictic lakes", which circulate freely twice a year (in the spring and in the fall) and which are stratified in the summer and in the winter. This cycle is typical of many Canadian lakes, particularly small ones. There are several notable exceptions to this cycle of mixing. Many lakes do not experience spring mixing between spring ice-out and summer stratification. This occurs because surface currents are negligible during the time that surface water temperatures increase from below 4°C to greater than 4°C. A second exception is lakes (e.g., Lake Ontario) which do not freeze over in the winter. In these lake, vertical mixing occurs from fall overturn through to summer stratification although they may not mix to all depths continuously. A third exception is lakes which freeze over in the winter but which do not stratify in the summer, allowing vertical circulation from spring through the fall. These lakes, called cold monomictic lakes, may stratify in the summer, but only on a temporary basis. They are typical of arctic areas and high mountain areas.

A fourth exception are lakes in which stratification is not completely controlled by thermal effects, but is controlled in part or in total by the effects of dissolved solids (TDS). In some lakes, permanent stratification occurs because of permanent salty bottom waters. These "meromictic" lakes are discussed further in section 7. International examples include the Black Sea and the Dead Sea. North American examples include Green Lake, Sunfish Lake, and Pot Lake. In other lakes (such as in the Kootenay Mountains, B.C.), seasonal differences in concentrations of bottom dissolved solids control stratification during part of the year while thermally induced density effects control vertical mixing during other times of the year. The seasonal TDS differences are controlled by seasonal differences in TDS concentrations in inflowing river water.

During stratification, vertical transport occurs due to diffusion through the thermocline and due to entrainment of cold bottom waters into the epilimnion owing to vertical currents. Entrainment is episodic, being a function of wind-driven shear currents (see section 6.3: wind shear stress-lake circulation interactions). The rate of entrainment is quite small during summer stratification, (varying from a few centimeters to a few meters); it may be as important as or more important than diffusion during much of the summer (see Stauffer and Lee, 1973; Snodgrass, 1977).

For measurement of vertical circulation characteristics, variables describing vertical diffusion, vertical circulation rates and water column stability are used. For measurement of gross stratification characteristics, inspection of vertical temperature profiles over time may be used. As horizontal variations are often smaller than vertical variations, annual data from one station is often sufficient. Inspection for isothermal conditions, for thermocline depths over time and the mixing cycles outlined above permit determination of the frequency of overturn, the maximum depth of vertical overturn and the rate of thermocline erosion. There are at least two dangers to this procedure. A lake may not mix in the spring even though isothermal conditions are observed. In this case, use of seasonal data for a non-conservative material such as oxygen is mandated. Due to wintertime bacterial consumption of oxygen, the oxygen concentrations of lower waters decrease during inverse stratifiction. If springtime mixing does not occur, the oxygen concentration at the beginning of summer stratification is lower than if vertical mixing had occurred in the spring. Low oxygen concentrations during the initial period of summer stratification is then evidence for the lack of springtime mixing.

The second danger to inspection of temperature data is the possible control of vertical circulation by vertical variations in TDS. Inspection of TDS data, conductivity data or concentrations of a conservative ion (e.g., chlorides) should show a uniform vertical concentration throughout the year. If the vertical concentration is not uniform, then water density may be controlled both by temperature and TDS. In this case, calculation of vertical densities over time (using an equation of state relating density to temperature and TDS) is required.

The stability of stratification may be measured by a dimensionless number such as the Gradient Richardson Number (GRN) or the Brunt-Vasallis number (BV). They are defined as

$$GRN = \frac{-g\ (d\rho/dz)}{(du/dz)^2}$$

$$BV = \frac{g}{\rho}\ \frac{d\rho}{dz}$$

where g = acceleration of gravity (L/T^2)
ρ = density (M/L^3)
u = horizontal velocity (L/T)
z = depth (L)
$\frac{du}{dz}$ = change of velocity with depth (T^{-1})

The Gradient Richardson Number (dimensionless) is a measure of the ability of buoyancy forces (the numerator) to resist vertical mixing and of the ability of vertical turbulent eddies arising from shear stresses (the denominator) to cause vertical mixing. Typically, for GRN less than 0.25, the water becomes unstable allowing vertical mixing. The Brunt-Vasallis Number is a measure of the stability of the water column (formulated in units of a frequency, e.g., sec^{-1}).

The Richardson number is both a measure of stability and of the ability of the lake to resist vertical mixing; calculation of du/dz makes its direct use difficult. The Brunt-Vasallis Number is a measure of stability, i.e., a directional tendancy, but not of a lake's ability to resist vertical mixing. Its calculation is direct and easy.

Various estimates of diffusion coefficients through the thermocline exist; they vary from molecular ranges (0.0125 m^2/day for heat) to values characteristic of turbulent motion (0.5 m^2/day). The actual value depends upon the method of analysis, upon assumptions concerning actual processes included in the measurement, and upon the stability of stratification. It may be calculated by considering the change of heat content of the hypolimnion (e.g., Lerman and Stiller, 1969; Li, 1973; O'Melia, 1973; Powell and Jassby, 1974; Sweers, 1969 a, b, 1970; Turner and Krauss, 1967; Spalding and Svensson, 1976). In many cases, thermocline entrainment is included in the calculation making the diffusion

coefficient larger than it actually is. For small lakes, it has been demonstrated that consideration of entrainment gives a calculated diffusion coefficient of the order of molecular (Hesslein and Quay, 1973). Another measurement for vertical transport is its relationship to vertical stability. Relationships between vertical transport and stability have been sought by various investigators. In general, coefficients of eddy diffusion in the hypolimnion (Mortimer, 1942), of eddy diffusion in the thermocline region (Blanton, 1973; Snodgrass and O'Melia, 1975) and rates of thermocline erosion (Liklejma, 1979; Hansen, 1979) appear to be inversely related to mean lake stability. This work is rapidly evolving, but more is required before simple engineering relationships will be available as management tools. Due to the dynamic nature of surface currents and surface temperatures, models capable of predicting the extent of vertical stratification (e.g., Huber et al, 1972; Beak, 1980a) on a daily basis will probably continue to be the tool of choice for some time.

A peculiar subset of lakes are man-made lakes (reservoirs) in which inflow typically occurs to the surface waters and outflow from the bottom waters. For reservoirs which have withdrawal from the bottom, the analysis presented above is incomplete. The elevation difference between inflow and outflow causes vertical currents as well as horizontal currents. For such reservoirs, flow across the thermocline due to the inflow-outflow effects (vertical thermocline currents), thermocline erosion due to wind-driven surface currents, and vertical diffusion all cause transport across the thermocline. A "Reservoir Pecklet Number", Pe, may be used to compare the relative effects of "advection flow" and diffusion. The Pecklet Number is defined (Octavio et al, 1977) as:

$$Pe = \frac{Q\ Zo}{AE}$$

where Q is the flow rate through the outlet at a depth Zo (m^3/day)
A is the reservoir cross-sectional area at that depth (m^2)
E is the coefficient of vertical diffusivity (m^2/day)

If Pe >>1.0, then advection will dominate over diffusivity.

Dimensionless numbers similar to the reservoir Pecklet Number are in the development stage for comparing vertical diffusivity to thermocline erosion, the dominant processes in most lakes (BEAK, 1980 provides some of this development). Such numbers will have particular value for management of salt- or TDS-induced stratification. Dimensionless numbers for horizontal flow and for the effects of atmospheric processes on vertical stability are presented in sections 6.4 and 6.3, respectively.

2.1.1.3 Horizontal Circulation Characteristics

Possible Measurements

1. dominant horizontal currents
2. exchange rates between embayments and lakes
3. time-scale of achieving well-mixed conditions
4. chemical horizontal gradients

Horizontal currents and horizontal diffusion are the dominant mechanisms of horizontal transport in lakes. During stratification the largest currents are confined to epilimnetic waters; the currents in the deeper waters of the epilimnion are smaller than the surface currents due to frictional effects between vertical fluid layers. Some currents are found in the hypolimnion - these are generated either by inflow effects from river water (the inflow reaches the hypolimnion due to the high density of the revier water) or by the transfer of horizontal current energy from surface waters. Transfer processes include seiches and longitudinal tilting of the thermocline. Seiches result from the rhythmic displacement of water mass involving internal oscillations of an isotherm. A seiche is a long-standing wave with wave length of the order of basin dimensions.

For a lake, small currents become eddies as the size of the water region being considered increases. Consider an example (Fig.2.1.1(a)). A current is a mean flow across a cross-sectional area. consider the sketch shown. the Movement of water across planes A-A^1 and B-B^1 are currents - mean motion across the given cross-sectional area. However, across plane C-C^1, the mean flow has a value of zero; hence, there is no current. Rather, the observed transport motion is an eddy and for the physical scale of the whole lake, transport would be termed horizontal eddy diffusion. The same example is relevant for differentiating transport due to currents from transport due to eddies for much smaller sized areas. This is the reason why it is observed that the eddy diffusion coefficient increases with the portion of the lake area being considered (e.g., an increase in length scale). For this reason, more transport is attributed to diffusivity processes and less transport is attributed to advection as one increases the portion of lake area being considered (i.e., as the length scale or the area scale is increased). The relationships of diffusivity processes to length scales are typically formulated as power laws such as the four-thirds power law (diffusion coefficient varies as the length scale to the four-thirds power). Such laws are based upon both theoretical and empirical considerations. Some care, however, must be used as the exact relationship will change from length scale to length scale.

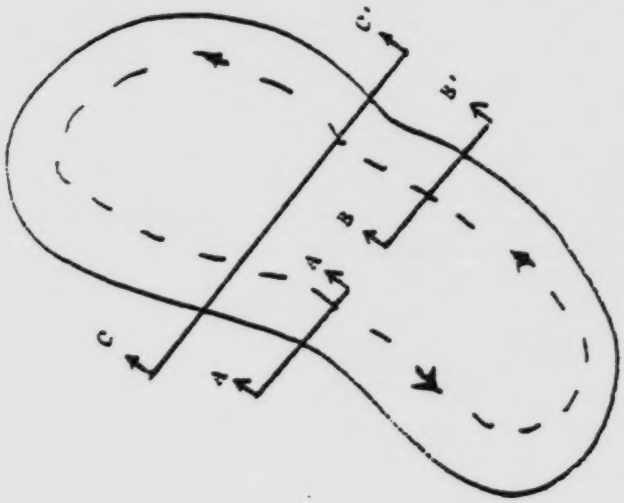


Figure 2.1.1(a): Comparison of advection and diffusion

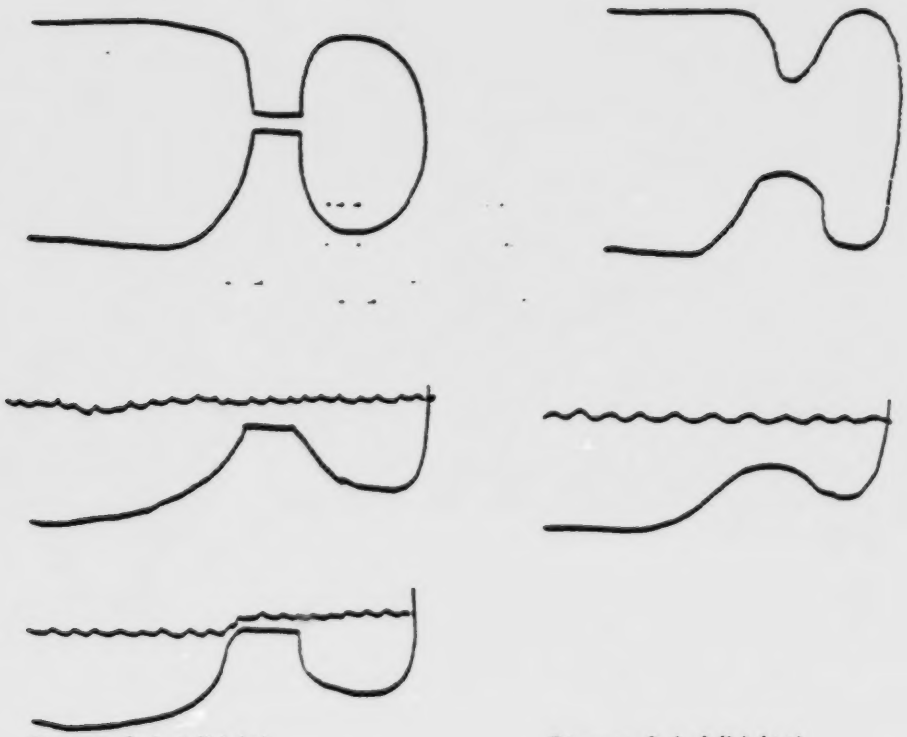


Figure 2.1.1(b)(i)

Figure 2.1.1(b)(ii)

Figure 2.1.1: Sketch of various lake-embayment systems

These descriptions provide the background for calculation of horizontal currents and exchange rates between embayments and lakes. Consider a lake to which an embayment is connected (Figure 2.1.1(b)); this example is pursued with the management option in mind that certain situations may exist in which disposal of tailings in embayments is feasible). Figure 2.1.1(b)(ii) shows an embayment connected to a large lake by a narrow neck of land. In the top cross-sectional sketch, the embayment water level is controlled by the lake; the embayment water level is controlled by the hydraulic flows over the lip in the bottom cross- sectional sketch.

In the lip controlled example, flow is normally from the embayment to the lake; occasional flow may occur from lake to the embayment if the lake level rises sufficiently. In Figure 2.1.1(b)(i), flow will occur from the lake to the embayment if lake surface currents are sufficiently strong; however, a limit is reached beyond which no more flow can occur into the embayment. When lake currents decrease or shift direction, flow may occur from the embayment to the lake. If weak stratification occurs in the channel, inflow to the embayment can occur in surface water with a simultaneous return flow in lower channel waters. For Figure 2.1.1(b)(ii), inflow may occur along one shore while outflow occurs along the second shore simultaneously.

These examples point out the complexity of horizontal flows between lakes and embayments; this makes their measurement quite difficult. Measurement can be achieved by employing a string of current meters at appropriate locations. The current meters give direction and velocity of flow. But for many applications, such as for assessing gross lake characteristics, use of an "exchange rate" is sufficient. An exchange rate is the cumulative volume of water which enters the embayment over a period of time (units, m^3/day or km^3/year); the same volume of water leaves the embayment as entered it in that time period if the lake surface remains at the same elevation over time. In additon, there would be an extra flow leaving the embayment if there is inflow to the embayment from groundwater or from surface runoff. This outflow is further explored in Section 5.2 (Drainage Basin-Lake Interactions).

To measure horizontal circulation, empirical observations with drogues and dye give general speed and direction for the time period of measurement.

Current meters allow precise measurements of current speed and direction. Dye experiments permit determination of diffusion characteristics. However, both current and diffusion data are generally only valid for the periods of measurement; extrapolation to an annual period is

often difficult. Use of mathematical models of lake physics permit calculation of dominant currents and diffusion characteristics over time; field data is needed for determination of unknown model coefficients. Such models permit calculation of the timescale to achieve well-mixed conditions. In small lakes, this time scale is relatively short, while in Lake Superior, it is estimated that a few years are required for shore water to uniformly mix with off-shore waters. Since lake physics are fairly well described mathematically, use of such a model in conjunction with the gathering of field data is often the method of choice.

To measure exchange rates, dye, drogues and current meters may be used together with mathematical models for lake currents. However, the amount of data, manpower, and computational effort required may not be justified for a management objective of determining gross lake characteristics. If it is assumed that average circulation patterns on a monthly basis are desired, then monthly exchange rates are sufficient. For the example in Figure 2.1.1(b)(ii), use of one water level recorder in the embayment and one in the lake, precise knowledge of channel geometry, and the assumption that the embayment acts as a Helmholtz resonator (see Freeman et al, 1974) should be sufficient to calculate such exchange rates. For the example in Figure 2.1.1(b)(ii) use of water level recorders in the embayment and at a few lake locations, and use of a method such as that of Defant (see Mortimer, 1979) would permit calculation of exchange rates. The resonator method assumes that the surface level of the embayment rises and falls in response to exchange flows such that the embayment water surface is constantly horizontal (an adequate assumption for transport through narrow channels) while a method such as that of Defant is used where the surface water level does not remain horizontal, but rather tilts in a significant fashion due to horizontal movement of a surface gravity wave (surface seiche).

If there is a significant input of a contaminant or other chemical element to the embayment from a river, the concentration will decrease from the source to the embayment-lake narrows. The exchange rate can be calculated as a function of the concentration gradient and input rate of the element to the embayment.

2.1.1.4 Perspective on Lake Physics

The effects of interactions between lake physics and lake sediments are examined in section 2.2, between the drainage basin and lake in section 5.2 and between the atmosphere and the lake in section 6.3. The relative importance of vertical and horizontal characteristics is examined here.

For the deep lake disposal option, two particular questions merit consideration. How often does a lake achieve well-mixed conditions? How efficient is the lake at transmitting the energy of large surface currents to bottom water?

Due to the heating-cooling cycle, the majority of Canadian lakes completely mix top to bottom at least once per year. Hence, lakes in which thermal effects control vertical density differences achieve well-mixed conditions on a timescale of once per year. Thus, on the average, any radionuclide or other soluble element which reaches the water column would be uniformily distributed on an annual basis, provided it is not affected by biota or by chemical reactions. For time scales of 100 years and greater, these lakes may be considered as well-mixed.

For embayments, a key parameter is the ratio of the time required for the embayment to achieve conditions similar to the lake to the time for the lake to achieve well-mixed conditions. Formulation of a parameter for assessing this ratio has been initiated by BEAK but requires more development and testing. Such a parameter examines the significance of horizontal lake processes relative to vertical lake transport processes upon embayment hydrodynamics.

With respect to vertical transfer of horizontal current energy, the efficiency can be determined in a quantitative fashion using a mathematical model of lake hydrodynamics. The transfer is a function of the resistance of water to motion (friction between vertical layers). In a uniform fluid with friction, the transfer is less than 100 percent. Under stratified conditions, buoyancy forces act to significantly protect bottom waters from large surface currents. Since the transfer efficiency is dependant upon the time of the year, an appropriate management parameter might consider the probability of a surface current of a certain size occurring in a given season and the stability of stratification in that season if one desires to avoid use of a mathematical model. Such a parameter is not presently available to our knowledge.

2.1.2 Water Body Chemistry

Water body chemistry refers to the chemical concentration of the dissolved and particulate phases in the water column of a lake. Inferences about the importance of the measurements can be made only when such information as "background" levels and "biological response" to different concentrations are known. Measurements of concentration are for a given point in time. The long-term change can be estimated only when estimates of long-term changes in inputs of these elements to the lake have been made (see Section 5.2, 6.2).

In this report, biological regimes (section 2.1.3) and inorganic particulates (section 2.1.4) have been separated out as specific entities. For assessing the soluble and particulate concentration portions of a water column, care must be taken to use data normally available. If one is interested only in a soluble phase, the water sample should be filtered (or centrifuged, etc.) before being transported from the field if exchangeable ions on the particles desorb. Further, if a digestion procedure is performed before removal of the particulates, some fraction of an element associated with the particle (adsorbed to, coprecipitated with, and incorporated into a chemical lattice as part of the host phase) will be included in the measurement, depending upon the strength of the digestion procedure.

To measure inorganic particulates, chemical methods would be needed which separate organic particulates from inorganic particulates.

2.1.2.1 Chemical Characteristics

Possible Measurements

1. buffering system controlling the pH
2. oxygenation characteristics
3. POC and DOC characteristics
4. major cations and anions
5. background radioisotope regimes

The chemical characteristics of a waterbody provide indications of the water's visual appearance, of levels of contaminants which might pose a

danger to human or ecosystem health and of the dominant geochemical mechanisms in the surface waters of the drainage basin.

A master variable of water is its pH. This allows inferences about many chemical and biological properties. For the majority of lakes, though, the pH is actually controlled by the inorganic carbon buffering system. For such lakes, the pH may be calculated if the TIC (total inorganic carbon concentration) and the alkalinity are known by using ionization constants relating the concentration of carbon dioxide to bicarbonate, and of bicarbonate to carbonate. Between the pH's of 6.5 and 10, most of the total inorganic carbon is in the form of bicarbonate ion. If the bicarbonate concentration is 10^{-4} M and the pH is 7 (i.e., $[H^+] = 10^{-7}$ M), then there is 1 000 times as much H^+ complexed with CO_3^{2-} as there is free (in the form of H^+, the form measured by a pH meter). Accordingly, the long-term behaviour of H^+ is highly influenced by the concentration of CO_3^{2-}.

The control of the bicarbonate system results from the fact that carbonic acid acts as a weak acid: - it dissociates only slightly to release small amounts of H^+. This compares to sulfuric acid, a strong acid which dissociates almost completely. If natural waters were composed only of strong acids and bases (e.g., H_2SO_4, NaOH), then H^+ in solution would control itself. Since the majority of H^+ in solution is associated with weak acids, then one must analyze the long-term behaviour of these weak acids to determine the long-term pH.

To assess present conditions, experimental measurement of a titration curve (in a closed laboratory titrator) to beyond the carbonic acid endpoint (pH 4.2-5.7 depending upon the concentration of TIC) and comparison with the curve for carbonic acid is one method which allows statements concerning the degree of control exerted by the TIC system. If a substantial departure is noted, either experimental error has occurred or else another weak acid is controlling. Candidate acids include ammonia, iron hydroxide complexes, hydrogen sulfide and weak organic acids. Generally, for most surface waters, there is no evidence to suspect that carbonic acid is not the controlling system. The other acids are particular candidates in sediments, groundwater, and bogs, etc.

Accordingly, for the long term, one needs statements about long-term variations in the bicarbonate system in order to predict the long-term pH. This is especially true if increasing lake acidification occurs.

Oxygenation characteristics are a measure of the dominant biological communities present. Typically during stratification, the bottom waters approach anoxic conditions. If anaerobic conditions occur, metals, sulfides and nutrient species (e.g., ammonia, phosphorus) which are more mobile under reduced conditions) will be transported into bottom waters. Lake overturn could transport these materials into the surface waters from which they could be discharged to downstream waters.

POC (particulate organic carbon) and DOC (dissolved organic carbon) characteristics are measures of the potential for oxygen decay through bacterial attack of the organic carbon. The origins of this material (algal production, woodland detritus) give an indication of the potential for decay. Their relative ratio and adsorption or complexing capacity provides information about their ability to remove radionuclides from a soluble form in the water column. Organics in the form of large molecules control the "colour" of many waters.

Major cations and anions present give general information concerning the potential for biological growth and evidence for the major weathering reactions in the watershed. The background radioisotope regime provides information concerning existing effects of weathering reactions upon such nuclides and gives indications of probable future regimes (without tailings present in the lake system).

2.1.2.2 Relevant Biological Growth Factors

Possible Measurements

1. nutrient (N, P) regimes
2. essential trace metals
3. vitamins
4. toxic trace metals and controlling organic compounds

For algal growth, levels of inorganic nitrogen (NH_3, NO_3), phosphorus, total inorganic carbon, essential trace metals and vitamins all influence the rate and extent of growth. The trace metals (e.g., Cu) can become toxic at elevated levels to both algal growth and to other ecosystem growth. Toxic effects can also occur due to entry of organics such as halogenated hydrocarbons into the lake.

On a priority basis, nutrient concentrations and trace metal levels are particularly important indicators of biological growth factors.

2.1.2.3 Perspective on Water Body Chemistry

Water body chemistry is a measure of the interactions within the lake due to weathering reaction inputs from the watershed, atmospheric inputs, and chemical-biological-physical removal processes within the lake. It is a measure of background concentrations of various metals (e.g., radium, resulting from weathering in the watershed). Various chemical elements are useful measures of the potential growth of various biota.

To predict future changes in water chemistry, one must able to predict firstly changes in inputs from the drainage basin (see section 5.2, 6.4) and secondly, changes in removal rates (see section 2.1.4, 2.1.5, and 2.2) for most elements. An exception is a parameter like pH for which one must be able to predict changes in TIC and alkalinity and whether there will be a change in the chemical buffering system controlling pH to a system other than the TIC system.

2.1.3 Water Body Biology

The biological community is an integral part of the freshwater aquatic ecosystem. Uptake of contaminants by organisms influences processes of sorption-desorption. Furthermore, biological processes oppose the establishment of geochemical equilibria in aquatic environments. An understanding of the characteristics of biological communities in proposed disposal lakes is essential to provide basic background data against which changes due to tailings disposal can be measured.

2.1.3.1 Biological Components

In this system, the major biotic components are described and then measurements of their characteristics are discussed.

Aquatic ecosystems are composed of the following major biotic components:
- microbes (bacteria, yeast, fungi)
- primary producers (phytoplankton, photosynthesis, aquatic plants)
- zooplankton
- benthic macroinvertebrates
- fish.

Microbes

Bacteria are responsible for organic decomposition and for cycling of various elements in aquatic environments. A variety of species are responsible for mediating various biochemical transformations in biogeochemical cycles. Culturing and identification of bacteria is extremely selective for certain types and these techniques tend to underestimate densities of heterogeneous species by factors of 10^3 to 10^5 (Wetzel, 1975). The process of epifluorescence microscopy has recently been developed for the direct determination of bacterial biomass (Francisco et al, 1973). For the purpose of defining the role of bacteria in lakes, and implications they have in the disposal of uranium mine tailings, it is likely more feasible to monitor the end products of bacterial metabolism (e.g., Eh, O_2) rather than to monitor the bacteria themselves. Thus, in this section we will briefly summarize the role of microbes in aquatic ecosystems without further describing how measurements of specific organisms can be made.

Microbial biomass is usually proportional to organic loading and to primary productivity, although numbers decrease markedly in acidic, dystrophic lakes having high concentrations of humic substances. Heterotrophic bacteria metabolize organic matter produced within

the lake or imported to the lake from other environments. Since organic matter is extremely important in the biogeochemical cycling of trace contaminants (e.g., Section 1.3.2), then the role of bacteria in regulating the organic matter pool has important implications in lakes used as disposal sites for uranium mine tailings. For example, microbes are responsible for the decomposition of higher molecular weight, organic acids to smaller, more soluble and mobilizable forms (Kusnetsov, 1975). Since trace contaminants are associated with the larger less mobile molecules, decay of the organics makes the trace contaminants more mobilizable. Also, because bacteria compete with abiotic particulates for trace metals (e.g., Ramamoorthy et al, 1977), they directly speed up the cycling of metals.

The sulphur cycle of lakes is regulated by bacteria. Protein-decomposing bacteria, primarily belonging to the genus Protens, produce H_2S as an end product in the metabolism of detrital proteins. Also, sulphate-reducing bacteria reduce sulphate, sulphite, thiosulphate, hyposulphite and elemental sulphur to H_2S by using sulphur compounds as hydrogen acceptors in oxidative metabolism under anaerobic conditions. This sulphide is then available for binding trace metal contaminants from sediment or tailings pore waters or from anaerobic hypolimnetic water to form insoluble metal sulphides. Sulphur-oxidizing bacteria oxidize sulphide to form elemental sulphur. This process potentially competes with trace metals introduced in mill tailings for H_2S. Photosynthetic bacteria (green and purple sulphur bacteria) also compete for H_2S which is oxidized to sulphate or elemental sulphur during the photosynthetic reduction of CO_2. This process would only be significant where significant portions of the euphotic zone are anaerobic.

The cycling of iron and manganese is dictated largely by reduction-oxidation conditions in lakes. Bacterial and photosynthetic metabolism greatly influence redox conditions, and thereby indirectly regulate the cycling of these metals (see Sections 1.3.2 and 2.1.2 for discussions of the role of Eh in biogeochemical cycles). The direct utilization of iron and manganese by bacteria in energetic transformations occurs in lakes but is minor in comparison with the heterotrophic metabolism of organic substrates in most natural conditions. The reader is referred to Wetzel (1975) for a more detailed discussion of bacterial transformations of iron and manganese. While metabolic transformations of these elements may be quantitatively minor components in their biogeochemical cycles, they are worth mentioning at this point due to the important roles played by iron and manganese in governing the biogeochemical behaviour of other elements which may be introduced by uranium mill tailings (e.g., Sections 1.1, 1.3.2).

Nitrogen, as one of the major constituents of cellular protoplasm, is a major nutrient which affects the biological productivity of aquatic ecosystems. Since biota are extremely important both directly and indirectly in the biogeochemical cycles of contaminants introduced to lakes, then the role of bacteria in controlling the availability of various nitrogenous compounds is of interest.

Elemental nitrogen in the form of N_2 is fixed from gaseous form to organic forms by blue-green algae and by certain bacteria. However, little quantitative information on nitrogen fixation by bacteria is available. Available data suggest that nitrogen fixation by heterotrophic bacteria is several orders of magnitude less than by blue-greens (Wetzel, 1975).

Ammonia is generated by heterotrophic bacteria as an end product of the decomposition of nitrogen-containing organic matter. A high concentration of ammonia in lakes is indicative of a high rate of organic deposition or of cultural pollution. This is particularly true where the hypolimnion becomes anoxic and a considerable amount of ammonia diffuses into the hypolimnion from the sediments.

Nitrification is broadly defined as the biological conversion of organic and inorganic nitrogen compounds from a reduced state to a more oxidized state under aerobic conditions. The process goes through a series of oxidation stages from ammonia (NH^+_3) to nitrate (NO_3); it provides the energy required for the reduction of CO_2 to organic matter in anaerobic environments. Organisms participating in these reactions include bacteria and fungi. Detailed reactions are discussed by Kuznetsov (1970).

Nitrate is reduced to N_2 by bacterial denitrification. In eutrophic lakes, denitrification influences the vertical distribution of nitrate. Bacterial denitrification is the biochemical reduction of oxidized nitrogen (primarily nitrate) to N_2 in the oxidation of organic matter. While denitrification occurs both aerobically and anaerobically, the process occurs most intensely in anaerobic environments such as in the hypolimnia of eutrophic lakes and in anoxic sediments. Processes of nitrification and denitrification can occur simultaneously in the same aerobic water mass, provided reduced microzones are available for denitrifiers.

Primary Producers

Primary producers in aquatic ecosystems include photosynthetic bacteria, phytoplankton (planktonic algae), aquatic macrophytes (aquatic weeds and larger species of filamentous algae) and attached or periphytic algae.

i) Phytoplankton

Phytoplankton populations undergo rapid changes in species composition and biomass in response to natural or man-induced environmental changes (e.g., additions of contaminants, acidification) and community succession occurs rapidly (nutrients, in the order of days or weeks). Phytoplankton communities should be monitored in lakes proposed as uranium mill tailings disposal sites, since they provide good indicators of overall environmental conditions and provide an important means for adsorption and sedimentation of contaminants released from mine tailings.

Possible Measurements of Phytoplankton and Ecological Significance

1. species composition in water
2. species composition in sediment cores
3. biomass
4. primary productivity
5. particle-size distribution
6. potential for food-chain transfer of contaminants.

A great deal of descriptive work has been devoted to the associations of algal species in various freshwater environments. While many species show wide tolerances to environmental conditions, nonetheless certain characteristic phytoplankton species associations occur repeatedly in lakes of similar environmental quality. Some of the major associations in lakes of similar trophic status are described by Hutchinson (1967).

Phytoplankton are routinely sampled by collecting 500 to 1 000 mL of lake water for each sample. Samples are collected at various intervals throughout the year and especially from the spring turnover to the fall turnover. Samples may also be collected from various depths to provide an indication of vertical zonation. After collection, samples are preserved with Lugol's iodine solution. The identification and enumeration of phytoplankton is then conducted by a qualified taxonomist using microscopic techniques, preferably the Utermohl method.

A number of phytoplankton indices have been developed to describe environmental conditions such as trophic status, degree of acidification, etc. (Thunmark, 1945; Nygaard, 1955). These indices are based on species associations and are useful for describing overall environmental conditions in lakes and may, therefore, prove useful for assessing the suitability of lakes for tailings disposal.

The composition of fossil phytoplankton in lake sediments may be used as "fingerprints" of historical environmental trends in lakes. For example, BEAK is presently conducting a study of diatom remains in Shield lakes showing varying degrees of acidification. Recently, Haworth (1980) showed a very good relationship between the long-term planktonic record and the fossil diatom record in a small lake. The long-term fossil record can, therefore, define environmental trends in the past which may be used to make projections of future trends. While these techniques have been used successfully, it is generally recognized that most information about the historical record is obtained from an interdisciplinary study of several parameters such as fossil diatoms, pollen, radioisotopes, etc. Reliance on one parameter may lead to erroneous conclusions.

Phytoplankton biomass or standing crop can provide a measure of the ability of phytoplankton to "strip" the water column of trace contaminants through the processes of adsorption, death and sedimentation. This stripping process should be directly related to phytoplankton biomass. Biological uptake by algae is described elsewhere (Section 2.2). Algae have been used to treat trace metal contaminated lakes including lakes receiving mining wastes by adsorption and settling (e.g., Jackson, 1977, 1978; Gale and Wixson, 1979). Jackson (1977, 1978) reduced trace metal concentrations in lake water by stimulating the growth of algae through fertilization with sewage.

Density of organisms is a commonly used measure of phytoplankton standing crop. Density data are converted to biomass by measuring average cell dimensions and calculating cell volumes, assuming a specific gravity of 1.0. This type of analysis is normally performed by phytoplankton specialists in conjunction with taxonomic analyses. Biomass can also be determined gravimetrically, preferably using lyophilization (freeze-drying).

Primary productivity of phytoplankton in lakes has received an extraordinary amount of attention in limnology. Primary productivity is not necessarily proportional to biomass due to the complicating factors of zooplankton grazing and settling of dead and living cells. Measurements of primary productivity have been made for a large number of aquatic systems, and there is no shortage of data for comparative purposes. Conceptually,

the rate of sedimentation of trace contaminants due to settling of phytoplankton cells and extracellular products should be proportional to the primary productivity rate. Eutrophic lakes with high productivity rates may be more efficient at removal of contaminants from the food chain, since the proportion of phytoplankton consumed by higher trophic levels generally declines with eutrophication and consequently the uncycled portion sedimenting also increases.

Considerable effort is required to give a good estimate of production rate. Water samples are collected from several depths in the water column and returned to the depth of collection in "light" and "dark" bottles. These bottles are clear, colourless, glass-stoppered BOD-type bottles with an opaque covering on the dark ones to prevent light penetration. To measure the amount of inorganic carbon fixed to produce organic carbon, photosynthesis is monitored by observing the evolution of O_2 or the uptake of C-14. Dark bottles are used to measure community respiration while light bottles measure net productivity, or photosynthesis minus respiration. Incubation times in the water column vary, depending upon C-14 uptake or an O_2 production rate. Measurements are generally repeated many times to account for meteorologic and temporal variations in order to provide a good mean estimate. Productivity is generally expressed on a per unit surface area basis and is corrected for a 24 h period. Details on measuring primary productivity are provided by several papers (e.g., Schindler and Holmgren, 1971). Recently more rapid incubator techniques have been described by Fee (1980).

While the effort required to provide good productivity estimates is considerable, productivity would undoubtedly be a useful tool to assess overall trophic status and to monitor the impact of tailings disposal. However, from the point of view of screening lakes, standing crop, species composition and selected nutrient measurements allow a good estimate of productivity.

ii) Macrophytes

Macrophytes are important in lake systems where shallow littoral areas are abundant, but comprise only a minor portion of the total biomass of primary producers in deep lakes with steeply-sloping sides. Macrophytes require adequate light penetration to the sediment, suitable sediment for rooting and some degree of protection from wave action. Environmental requirements vary considerably among plant species.

Where littoral areas comprise a significant portion of a lake, biogeochemical relations of contaminants in sediments will be affected by the plants. Macrophytes remove trace elements from sediment and water (e.g., Mayes et al, 1977; Mudroch and Capobianco, 1979), although some plants may accumulate from water while discriminating against uptake from sediment (Mortimer, 1977) and other may accumulate from water and transfer contaminants to sediment (Mortimer and Kudo, 1974). Other species such as some Nuphar species have been shown to 'pump' large volumes of air through rhizomes deep within the sediment and so affect local redox potential and provide a transport route for radon directly to the atmosphere (Dacey, 1980).

Possible Measurements of Macrophytes and Ecological Significance

1. areal extent of littoral zones
2. species composition
3. potential for macrophyte grazing by animals

The measurement of the areal extent of littoral zones in lakes proposed as disposal sites would provide an estimate of the relative importance of aquatic macrophytes in biogeochemical cycles of the lake. Macrophyte zonation will provide clues on the deposition-erosion patterns which will influence contaminant accumulation in shallow environments around the lake. Areal mapping and airphoto analysis, with records of the period of peak summer development of macrophytes, would be required for these assessments. From such mapping, initial predictions can be made of the potential amount of grazing by animals.

Species composition is measured by field survey. Transects through littoral zones should be established, based on community variations observed in aerial surveys. Variations in species composition and density, with depth and sediment changes, should be determined by a qualified aquatic botanist. As various species have specific environmental requirements, environmental conditions in the littoral zone can be assessed. These results can be interpreted to define patterns of erosion, wave action and sedimentation which would influence the behaviour of sediment-bound contaminants.

iii) Zooplankton

Zooplankton in lakes includes all small planktonic animals occurring in the water column and is comprised of rotifers and two subclasses of Crustacea - the Cladocera and the Copepoda. These animals rely largely upon phytoplankton as a food source.

Like phytoplankton, zooplankton communities respond relatively rapidly to environmental change (e.g., contamination, acidification, temperature) and thereby provide a measure of overall environmental quality. Furthermore, zooplankton adsorb contaminants directly or accumulate them through the consumption of phytoplankton. The sedimentation of faecal pellets and shed carapace material formed from zooplankton remains is another means of demobilization of dissolved contaminants.

Possible Measurements of Zooplankton and Ecological Significance

1. species composition in the water column
2. species composition in sediment cores
3. density/biomass in the water column
4. particle size distribution
5. potential for food-chain transfer

Most work on zooplankton ecological relationships has been related to crustacean zooplankton rather than to rotifers or to ciliate protozoa. Some workers have related changes in zooplankton populations in the Great Lakes to cultural eutrophication (Brooks, 1969; Wells, 1970; Gannon and Beeton, 1971). Decreases in the size of individual zooplankton with eutrophication has been attributed to predation pressure from planktivorous fish (Wells, 1970) or to a greater abundance of small food items in eutrophic waters (Gliwicz, 1969). Patalas (1972) showed that cladocerans and cyclopoids are more abundant than calanoids in eutrophic lakes, while the reverse is true in oligotrophic lakes. Yan and Strus (1980) recently discussed responses of zooplankton in Shield lakes to acidification. Work on rotifers and ciliate protozoa has been underemphasized; protozoa offer an important food chain component, feeding on bacteria and small algae.

Various techniques are used for collecting zooplankton. "Wisconsin" nets made of fine mesh are used to take vertical hauls through the water column. Zooplankton traps are used to enclose a volume of water at a given depth. This water is then filtered through a fine mesh to concentrate the organisms. Zooplankton samples are usually preserved after collection using 4% formalin or formalin-sucrose solutions. A zooplankton expert would then conduct the identification using taxonomic keys and microscopic techniques and a qualified biologist would be required for data interpretation.

Among crustaceans, cladocerans usually predominate in fossil form in the sediments. There are many examples of the usefulness of cladoceran remains in interpreting historical

environmental conditions (Frey, 1969, 1974). Goulden (1969) used fossil chydorid cladocerans to assess the stability and responses of communites to environmental disturbances. Cladoceran remains have been used to show ecological shifts, agricultural disturbances, climatic change, shifts in productivity between planktonic and littoral sources and to interpret past changes in water level and littoral development (Wetzel, 1975). The historical record could be examined in proposed tailings disposal lakes to provide information on historical trends which may be used to predict expected stability or change in the lake following disposal.

Sampling of sediment must be done carefully to ensure that core collection does not disrupt the vertical zonation of sediment. Various devices are available for taking sediment core samples. Care must also be taken to ensure that the area sampled shows no evidence of hiatus or slumping which would disrupt the sedimentary record. Analysis and interpretation of samples would be carried out by specialists in zooplankton taxonomy and biology.

The density of zooplankton is determined in conjunction with identification to provide a measure of relative and absolute abundance. Overall crustacean zooplankton abundances are generally higher in eutrophic lakes than in oligotrophic lakes. Since zooplankton along with other biological and abiotic particulates mediate in the removal of dissolved and particulate forms of contaminants from the water column, the zooplankton densities would be directly related to their capacity for stripping the water column of contaminants originating from mine tailings.

Biomass of zooplankton could be measured gravimetrically by filtration, but data on biomass in the literature is much less extensive than data on density (see Bottrell *et al*, 1977).

iv) Benthic Macroinvertebrates

Benthic invertebrates include all visible animals without internal skeletons living in or on the substrate. Benthic communities reflect overall environmental quality since environmental tolerances vary with each species. Unlike phytoplankton and zooplankton communities which may change rapidly due to natural environmental factors, the benthic

community is relatively stable on a seasonal basis and immobile so that responses to environmental disturbances such as pollution can be readily assessed. Furthermore, benthic animals accumulate trace metals from the environment and influence sediment-solute contaminant distributions by bioturbation.

Possible Measurements of Benthic Macroinvertebrates and Ecological Significance

1. species composition
2. species composition in sediment cores
3. density or biomass
4. bioperturbation potential
5. depth of occurrence in sediments

The benthic community is composed of invertebrates such as worms, clams, snails, crustaceans and assorted insects. Since various species and groups show unique tolerances of environmental conditions and considerable information has been compiled on these tolerances, biotic indices of environmental quality, based on benthic species, are widely used to provide quantitative measures of environmental contamination. Unfortunately, most of these biotic indices such as the Trent index (Woodiwiss, 1964), the Beak index (Beak, 1964) and the arthropod index (Hilsenhoff, 1977) are most useful for describing the effects of organic pollution and are less sensitive to the effects of trace metal and radionuclide contamination. Nevertheless, species reductions in some freshwater environments have been attributed to metal contamination (e.g., Guthrie et al, 1974; Cherry et al, 1979; Occhiogrosso et al, 1979; Moore, 1979). A knowledge of species present in lakes proposed for tailings disposal is necessary to provide information on overall environmental conditions (e.g., trophic status, acidification, etc.) and to provide baseline data for biological monitoring. Indeed, if only one biological community is to be monitored, the benthos should be selected owing to the wealth of information available on species tolerances.

Benthos in unconsolidated sediments may be collected using bottom grab samplers, however, most benthic workers now agree that cores are more quantitative than grab samples for estimating macroinvertebrate densities. Quantitative samples are routinely taken using a Ponar grab or an Ekman grab. Sampling benthos from hard substrata is more difficult and usually requires the use of scuba and either an air-lift sampler or a suction dome sampler. After collection, samples are sieved to remove fine sediment particles and the animals and coarse debris are preserved in 70% ethanol or in 10% formalin.

In the laboratory, samples are sorted from debris into major taxonomic categories, are identified to genus or species using taxonomic keys, and then enumerated.

The fossil midge (Chironomidae) family in the sediment provides long-term historical information on the evolution of a lake environment. Midges are generally very abundant in lakes and their hardened head capsules are frequently well-preserved in lake sediments. Thus, as a lake becomes more eutrophic with decreasing hypolimnetic oxygen concentrations, a succession of midge populations from those requiring high oxygen levels to those tolerating decreasing oxygen content would be expected. Stahl (1959) demonstrated this type of succession in sediment cores. As in all sediment palaeolimnological studies, sediment cores must be taken very carefully in areas showing no evidence of hiatus or slumping in the past which would disrupt the sediment record. Collections of fossil midges should be identified and interpreted by qualified taxonomists.

Densities of benthic invertebrates in samples should be determined during the analysis of samples since relative and absolute concentrations of the various species present will provide the quantitive information necessary for the determination of various indices of environmental contamination. These values will also be useful to compare with other data on responses of benthic densities to contamination. For example, Moore (1979) showed that benthic densities were inversely related to sediment contamination by trace elements originating from mine tailings in an embayment of Great Slave Lake. Similarly, Wentsel et al. (1977) found that numbers of larval midges were inversely related to sediment metal levels in a contaminated Indiana lake.

While biomass estimates can be made gravimetrically with little effort, the data base in the published literature is much less extensive than density data, and it is our feeling that biomass estimates should be considered either as a minor component or should be ignored in environmental studies pertaining to deep lake disposal.

Rates of bioperturbation are perhaps best obtained through a combination of chemical means and species specific life history data relating principally to burrowing invertebrates. The degree of bioperturbation affects the magnitude of transport through sediments. The depth of their occurrence gives an indication of the zone of sediments or of tailings (if not covered by sediments) which will be affected by increased rates of transport.

v) Fish

The fish community presently existing or which could exist in lakes proposed as disposal sites should be considered as part of environmental study programs since fish can be used to indicate overall environmental quality and are known to influence the biogeochemical behaviour of trace contaminants. In addition, fish provide the food chain link between contaminants from tailings to man.

Possible Measurements of Fish and Ecological Significance

1. species composition and relative numbers
2. importance as a natural resource
3. potential for development of a fishery

Knowledge of the fish species present in a lake and potentially present after tailing placement is important because abilities of various species to accumulate contaminants vary and because only certain species are direct food and economic resources for man.

Species composition in lakes should first be determined from provincial government records (e.g., Ministry of Natural Resources in Ontario). Where inadequate data exists, fisheries surveys should be conducted to identify species present and critical areas (e.g., spawning and nursery grounds) in order to mitigate against major impacts of disposal if it is assessed as desirable to maintain the existing community. Various standard techniques such as gillnetting, trapnetting, electroshocking, sonar, etc., are available to fisheries biologists to assess fish communities.

Of prime concern is the importance of a lake as a fishery resource. Many lakes are economically important for recreational and often as commercial fishing. The potential impact of tailings disposal on fisheries must be considered when calculating the economic benefit of deep lake disposal of mine tailings. Knowledge of the form and bioaccumulation potentials of various components in mine tailings to important food chain species is required for assessing short- and long-term impacts.

Socio-economic studies including surveys of existing and potential fishing camps, outfitters and commercial catches would be required in this investigation. Potential impacts of disposal on contaminant levels in fish would be predicted by biologists using geochemical models of predicted environmental concentrations and knowledge of bioaccumulation factors.

Potential economic losses may then be determined. In such a case, the cost of a rigorous post-disposal program for monitoring contaminants in fish flesh should be part of any cost-benefit analysis of a deep-lake tailings disposal project.

2.1.3.2 Diversity and Stability of Ecosystems

Diversity is a measure of the richness of species in a biological community. In general, where environmental conditions are not highly variable but are more or less constant (i.e., in stable environments), the density of species present in various biological communities (microbes, phytoplankton, fish, etc.) is generally high. Thus, diverse communities are characterized by a high number of species present without a great over-abundance of any one species. Conversely, where environmental conditions are stressful, biological diversity is low with fewer species present and large numbers of individuals of a few tolerant types. Diverse communities tend to be resistant to severe biological change but are not resilient in the face of unusual environmental stress. The converse is true for communities of low diversity. Since the introduction of contaminants from uranium mine tailings may represent a significant environmental stress, certain measurements of biological diversity should be made in conjunction with environmental study programs associated with deep lake disposal.

Factors representing environmental stresses (e.g., contaminant concentrations, dissolved oxygen levels, etc.) are described in other sections of this report (e.g., 1.3.2, 2.1.2). Here we are concerned with biological consequences of environmental stability/instability.

Possible Measurements

1. Shannon-Weaver diversity index
2. Brillouin diversity index
3. production/biomass ratio

Many indices have been proposed to measure diversity (see Wilhm, 1967; Whittaker, 1972; Peet, 1974). Pielou (1975) discussed the theoretical basis for the use of indices from information theory to measure species diversity and has suggested conditions under which various indices are appropriate. One of the most commonly used diversity indices is the Shannon or Shannon-Weaver index (Shannon and Weaver, 1949). Kaesler and Herricks (1977) provided arguments favouring the use of Brillouin's (1962) index of diversity over that of Shannon for the purpose of applied ecology. Formulae for the Shannon and Brillouin indices are:

$$H = \sum_{i=1}^{s} \frac{Ni}{N} \ln \frac{Ni}{N} \qquad \text{(Shannon)}$$

$$H = \frac{1}{N} \ln \frac{N!}{N_1! N_2! \ldots Ns!} \qquad \text{(Brillouin)}$$

$$H = \sum_{i=1}^{s} \frac{Ni}{N} \ln \frac{Ni}{N} \qquad \text{(Approximation Brillouin Diversity)}$$

where s = number of species
N = number of individuals (total)
Ni = number of individuals in the ith species

Among aquatic biological communities, the published data base on diversity is greatest for benthic macroinvertebrates and effects of environmental perturbations on diversity are well documented.

In lakes where diversities are relatively high, it can be expected that the environment is relatively stable. In these situations, declines in diversity and subsequent species eliminations due to contaminant introductions will be most pronounced. On the other hand, lakes having low diversity and relatively few tolerant species will be more resilient to environmental contamination and biological communities may be more able to cope with environmental stresses associated with tailings disposal.

The productivity/biomass or P/B ratio is another measure of ecological stability. In environmentally stressful environments, the ratio of production (i.e., grams of tissue produced per unit time) to biomass (i.e., standing crop weight of tissue) is generally high. This is because environmental extremes do not permit the existence of large biomasses, but high rates of production are required to maintain biological populations. The opposite applies in environmentally stable environments. P/B ratios are used in ecological studies but, to our knowledge, have not been applied in environmental management. These ratios in proposed disposal lakes may compliment other information on biological communities collected during environmental survey work.

2.1.3.3 Perspective on Water Body Biology

An understanding of the biological components of lake ecosystems can provide environmental managers with as much or more information on the environment and on historical environmental trends than can an understanding of abiotic components.

Organisms in lakes govern biogeochemical cycles of certain key non-conservative elements which may, in turn, govern the behaviour of introduced contaminants. Every species has a unique tolerance for each environmental variable (e.g., oxygen, trophic status, acidification, contaminants). As integrators of overall environmental conditions, the status of biological communities provides many means of measuring environmental quality.

Biological "fingerprints" of past environmental conditions are provided in fossil forms in lake sediments. Thus, a record of the past environment is available for assessing potential future cycles (e.g., eutrophication, acidification).

Organisms regulate the mobilization and demobilization of contaminants in the aquatic environment directly (e.g., through bioaccumulation) and indirectly (e.g., by influencing Eh, sulphide, etc.) and, therefore, will affect the fate of elements introduced to lakes in uranium mine tailings.

Knowledge of biological conditions is also necessary to monitor environmental impacts of tailings disposal and to design mitigating measures to reduce these impacts.

The biota of a lake directly or indirectly influences the overall earth ecosystem of which man is a member. While direct effects such as losses of a fishery are more easily assessed, it is impossible to predict and to measure the overall ecological impact of deep lake disposal on man. Thus, we must endeavour to protect the biological communities of disposal lakes from detrimental effects of mine tailings in order to minimize the ecological consequences to all species including man.

2.1.4 Water Body Inorganics

Water body inorganics enter the water-column of a lake from a watershed or from the lake's sediments. While eroded bottom sediments may leave a lake by the outflow, their dominant tendancy will be to resettle after the erosional water energy is dissipated. The importance of eroded sediment particulates is as a site for uptake or release of various elements, particularly for heavy metals, and the subsequent settling of the particles.

Input rates and loss rates of inorganics are examined in section 5.1, 5.2, 6.2 and 6.4 and in sections 2.2 and 4.0 respectively; their in situ physical and chemical characteristics are examined here.

2.1.4.1 Physical Characteristics

Possible Measurements

1. settling rates
2. size distribution and mineralogy

Because inorganics have higher settling velocities than organics, they may influence the removal of radionuclides from the water column more than particulate organics despite their relatively lower concentrations.

The settling rates of inorganics are a strong influence upon potential removal of heavy metals from the water column. Knowledge of their size distribution and mineralogy is useful for drawing inferences about their settling characteristics and their chemical characteristics.

2.1.4.2 Chemical Characteristics

Possible Measurements

1. sources of inorganics
2. adsorption characteristics

The sources of waterbody inorganics often hold the key to their ability to take up heavy metals such as radionuclides. If the surface is coated, the inorganic will have different uptake characteristics than if the coating does not exist. If it comes from a groundwater discharge zone and was subjected to alternating wet-dry cycles, it may be coated by oxides and hydroxides of iron and manganese. If it originates from

the lake's sediment, its surface will be controlled by past redox behaviour. If it originates in marsh areas, its surface may be coated by organics. If the inorganic has been in the water column for some time, it may be coated by vibrols (small organic molecules). They may change its surface characteristics in a significant fashion. Adsorption characteristics are one measure of the inorganic's ability to remove metals from the water column. Co-precipitation reactions and complexation are others. Literature data is sparse but rapidly accumulating. Certain elements may be released if particulate concentrations are larger than equilibrium water concentrations. Experimental measurements upon the inorganics of particular watersheds may be necessary.

2.1.4.3 Perspective on Water Body Inorganics

Water body inorganics are one potential mechanism for the removal of radioisotopes and other elements from the lake water column through the processes of uptake (adsorption, precipitation, ion exchange, complexation) onto the particulate of a soluble contaminant and settling of the particulate to the sediments. Chemical characteristics control the potential uptake while physical characteristics control the rate of removal.

The actual concentration at any one point in time is highly dependent upon the rate of input from the watershed. Large inputs will usually be associated with meteorological extrema. If an upset of tailings were to occur due to winds associated with a rainstorm, it is probable that substantial erosion of particulates from watershed erosion would occur at the same time as the upset. This input represents an extra amount of inorganics available for capturing released contaminants. Such phenomena ensure that the lake has potential self-compensating mechanisms associated with certain kinds of tailings disturbances.

2.1.5 Interactions between Particulate and Soluble Phases of Water Body

The soluble and particulate characteristics of a water body have been considered above (2.1.1-2.1.4). The sources of these materials are considered below (5.0-6.0). This section considers those mechanisms which have the potential for changing soluble phases into particulate phases. This section is considered as a separate section because transformation of soluble elements leached from the tailings-sediment system to a particular phase is an internal self-compensating mechanism for minimizing the net amount of such elements reaching the surface ecosystem. Hence, the controls on mechanisms which transform soluble elements into particulate, potentially settleable phases are quite important.

To differentiate the soluble phase from the particulate phase, an operational definition, such as material which passes a 0.45 um glass fibre cellulose membrane as being soluble, may be used. Soluble and low molecular colloids pass the 0.45 um filter while high molecular weight colloids and particulates do not. Such a definition is useful for separating gross behaviour. Particulate materials settle whereas soluble materials do not settle if it can be assumed that intermolecular forces caused by Brownian motion dominate over the forces of gravity. This differentiation is made because soluble materials as large as 0.1 um in effective diameter have some tendancy to settle. Particulate phases may be divided according to whether they are organic, viable particulates (algae, plants, bacteria), organic non-viable or inorganic.

2.1.5.1 Biological Production Characteristics

Possible Measurements

1. standing crop of algae, zooplankton, rooted aquatics
2. rate of primary production of various plants
3. rate of secondary production
4. potential for food chain transfer

Trace metals may be taken up actively (as growth factors) or passively (due to their association with organic molecules or diffusion). Rates of production give some evidence for these mechanisms. The viability of the plants provides evidence for their ability to keep such metals within their cells. Research is needed to relate metal uptake to such characteristics. Contaminants may be transferred up the food chain, depending upon the association of the contaminant with cellular materials. Transfer into large biological particles provides for faster resettling of contaminants from the water column.

2.1.5.2 Biological Decomposition Characteristics

Possible Measurements

1. bacterial and fungal populations present
2. ATP concentrations
3. rates of oxygen consumption in the water column

Bacteria and fungi, in effect, make the global cycles of matter function. Without them, the world would be inundated with organics at a phenomenal rate. They are a key to the recycling of matter from par-

ticulate forms to soluble forms in the water column of lakes. Their importance to the whole lake ecosystem compared to that of phytoplankton has probably been significantly underemphasized in research work and in environmental decision making.

Accordingly, while a basic understanding of the ecology of bacteria exists, it has not been translated into a readily usable, standard set of parameters. In addition, other microorganisms are important in the breakdown of complex matter including yeasts, molds, protozoa and metazoa. Their importance cannot be minimized.

Bacterial numbers of major groups (e.g., heterotrophs, nitrifiers, sulfide oxidizers) provide some evidence of the decompositon characteristics of a water column. Unfortunately, numbers of bacteria (within the same order of magnitude) often do not show correlations with decomposition characteristics as measured by oxygen consumption rates. It is expected that higher bacterial numbers should produce higher decay rates. Why such correlations do not exist in certain studies is a matter of debate. Ideas suggested include the inability of measurement techniques (e.g., plate counts) to measure the viability of bacteria. More indirect measurements such as levels of ATP in the water column or sediments may be more appropriate.

The rate of oxygen consumption in the water column (analogous to BOD, but conducted for 1-2 days without the addition of nutrients) and the production of CO_2 are two measures of decomposition characteristics. The former can be calculated from monthly oxygen profiles in the lake hypolimnion, if there is insignificant transport of oxygen across the thermocline and insignificant production of sub-thermocline oxygen. The latter would need to be measured using laboratory procedures.

Rates of decomposition provide some evidence for the potential for removal of heavy metals from particulate organic material. It provides possible evidence for removal from inorganic materials because bacteria seek surfaces rather than floating freely as do algae. Evidence for unequal removal of heavy metals in comparison to oxygen consumption is needed before the use of decomposition rates as an indicator of dissolution behaviour of the metal could be rejected.

2.1.5.3 Ecological Relationships of Community Structure to Water Chemistry

Possible Measurements

1. sediment fingerprints due to previous changes in lake ecosystem
2. case studies from other lakes

In general, the biota of a specific lake is adapted to its chemical and thermal characteristics. Significant changes in the biota, induced by future changes in lake water chemistry may change the ability of the biota to act as scavengers for trace metals.

Three potential approaches to predicting the future biota are i) use of evidence from an examination of the lake sediments, ii) an examination of nearby lakes, and iii) literature case studies. If different biota are evidenced in the sediments (e.g., the presence or absence of diatom shells), it may be possible to correlate these shifts to past changes in water chemistry. Knowledge of drainage basin changes is usually essential, because past lake water chemistry is unknown. Evidence from nearby lakes which have changed in the past few decades may provide clues to potential changes. For example, evidence is accumulating that there is a shift in acidifying lakes from floating algal populations to mats of algae located on the sediments. Such large shifts allow inferences concerning gross characteristics for uptake of metals such as radionuclides. Literature case studies would provide similar evidence.

2.1.5.4 Other Uptake Mechanisms

Possible measurements

1. adsorption isotherms
2. complexing capacity
3. potential precipitates and properties

Adsorption, complexation and chemical precipitation are three mechanisms for uptake of radionuclides and other soluble contaminants in addition to biological ingestion. Algae, dissolved and particulate organics from woodlands, such as humic and fulvic acids, and inorganics such as silts and clays have different adsorption capacities. These capacities are a function of surface area, temperature, pH and other factors. In addition, humic and fulvic acids have large complexing capacities for heavy metals, depending upon the metal. Surfaces on inorganics may have sites for precipitation reactions. The interface between anaerobic and aerobic layers of water is the location of precipitation reactions for iron and manganese; such precipitation reactions form amorphorus precipitates and inorganic polymers which are capable of co-precipitating other heavy metals. The relative long-term importance of these mechanisms can be assessed by using present characteristics of existing lakes and projected changes in watershed behaviour.

Adsorption isotherms can be used to estimate adsorption behaviour quantitatively; experimental measurements between many trace metals and different surfaces are needed. The question of complexing capacity is a large area of research. Estimation of gross complexation characteristics is possible using procedures such as those of van den Berg et al (1979). Selection of various watershed organics, in addition to existing lake water organics, provides evidence for estimating future characteristics, assuming that these watershed organics will be eroded to the lake at some point in the future.

Use of a gross adsorption coefficient for a water is adequate for present lake conditions; the more detailed measurements are needed to allow inferences about future lake behaviour.

2.1.5.5 Other Dissolution Mechanisms

Possible Measurements

1. leaching measurements
2. redox reactions in anoxic water

In the oxic zones of most lakes, desorption reactions (the opposite to adsorption) will occur if particles settle into water which have significantly lower soluble concentrations and if the element is exchangeably adsorbed.

Redox reactions will cause dissolution if bottom waters are anoxic and if precipitates such as $Fe(OH)_3$ which are stable in oxic zones settle through these anoxic waters. Since these hydroxides of iron and manganese (found in lakes with high iron-clay contents for example) may have been formed at the aerobic-anaerobic interface, the net dissolution effect is a function of the rate of breakdown versus the rate of settling.

2.1.5.6 Bioaccumulation Factors of Radionuclide Concentrations in Aquatic Organisms

Possible Measurement

1. ratio of organism concentrations to water concentration of radionuclides

When quantifying the passage of radionuclides through terrestrial food chains, scientists generally express the magnitude of vectors of

passage in terms of transfer coefficients (e.g., Garner, 1972). The transfer coefficient is the relation between the concentration of a contaminant in one species or trophic level (feed) and the final flesh concentration in the species or trophic level that consumes it (meat), corrected for the daily intake of food by the consumer. This approach is logical if one assumes that the only means of entry of the contaminant into the consumer is through its feed. This assumption is usually taken as being valid for all but the first terrestrial trophic level (plants). For plants two pathways must be postulated: soil uptake and direct foliar deposition of airborne particles. In subsequent trophic levels direct uptake from the medium (air) is ignored and only dietary intake is considered.

In aquatic systems, however, the medium (water) is capable of holding large concentrations of contaminants in solution. Transfer pathway calculations must therefore consider not only alimentary assimilation of ingested contaminants, but also adsorption from the water. Indeed, Hall (1978) found that the final Ni-63 concentration in herbivorous zooplankton in a system with both dissolved isotope and isotope-labelled food algae was not significantly different from the final concentration in a system with only the dissolved isotope and no labelled food. The gill apparatus was identified as the primary adsorption site.

Partitioning contaminant uptake between the alimentation and adsorption pathways is prohibitively complex. A more realistic approach is to express concentrations at all trophic levels in terms of one basic concentration, that of the aquatic medium. This relation has been termed variously as "relative concentration" (e.g., Feldt, 1979), "bioconcentration" (e.g., Kenaga, 1980), or "bioaccumulation" (e.g., Mudroch & Capobianco, 1979). For the purpose of this document the term bioaccumulation factor (BF) will be used.

In theory, bioaccumulation is relatively simple to determine. Two quantities are required, the concentration of the element in the water and the concentration in the organism. Because of differential translocation, however, concentrations often vary from one organ to another in an animal (e.g., Hall, 1978), which can complicate calculations if the animal is too large to analyze entirely. Thus, some authors have calculated separate factors for fish flesh and fish bone (e.g., Anderson et al, 1963), while others have used whole fish values (e.g., Pradel and Zettwoog, 1977). For larger invertebrates, zooplankton, phytoplankton and macrophytes, whole organism values are usually given (e.g., Feldt, 1979).

Many different groups of elements in virtually all regions of the periodic chart have radioisotopes. Since different types of elements have different biological pathways and functions, it is not surprising that BF values vary from one element to another (see Table 2.1.1). In an attempt to order this disarray of factors, Amiard and Amiard-Triquet (1980) have grouped radionuclides according to their biological importance and periodic grouping. Table 2.1.2 summarizes their classification. Further refinement of this type of system may eventually facilitate predictions, in as much as results from experiments with one element may be applicable to calculations for other elements that are similar in biological and chemical behaviour.

Another fact evident from Table 2.1.1 is that BF values for a given element may differ greatly between authors. Many of these discrepancies may be due to methodological differences. Feldt (1979) found three experimental variables that had an effect upon observed BF values:

1. Concentration - the aqueous concentration in the system controls the BF because organisms have a limit to the amount of any element that they can assimilate and retain. When this limit is passed, the BF will decrease as the load increases because tissues will no longer accumulate materials.

2. Time - for any given aqueous concentration, there is a time delay before an element reaches its equilibrium concentration in the organism. For plankton, this delay is relatively short (0-2 days) but for fish larvae and adults it can exceed 40 days. Sampling before equilibrium is reached will produce invalid results.

3. Feeding regime - for fish, the BF response to a given aqueous concentration will depend upon the feeding habits of the species or, in a laboratory situation, upon the experimental feeding regime.

Another possible reason for discrepancies among field observations may be differences in the biology (species) or in the water chemistry of the different systems sampled. The behaviour of elements in well-buffered calcareous waters is often quite different from that in dilute acidic waters (Schindler et al, 1980). Since most BF values in the literature were derived in calcareous systems, there is a particular need for radionuclide BF data that are relevant to Shield waters.

Table 2.1.1 summarizes literature values for a few radionuclide-producing elements. The highest values in the table are all for aquatic plants. The capacity of macrophytes to assimilate large quantities of trace elements may well be related to their ability to mobilize these

Table 2.1.1: Some reported values for bioaccumulation factors (BF) in aquatic biota containing radioactive isotopes. BF equals conc. biota/ conc. water

Element	Fish (adult) Flesh	Whole	Bone	Invert.	Zoopl.	Aquatic Plants	Reference
U	10			100		1000	Thompson et al, 1972
	2			60		0.5	USNRC, 1977
					50		Pentreath, 1979
Th	30	150		500		1500	USNRC, 1977
					4000		Pentreath, 1979
Ra	3-4		100	500-1000		500-1000	Anderson et al, 1961
	1-14	2-11		200-10000		1000-10000	OWRC, 1971
	100-600	500-5000					Pradel and Zettwoog, 1977
	50			250		2500	USNRC, 1977
	7.4	54	60				EPS, 1978
					100		Pentreath, 1979
		30-60					CEA, 1979
Pb	100				100	200	USNRC, 1977
	0.5-150						Thompson et al, 1972
	9	275					IAEA, 1976
Po	500			20000		2000	USNRC, 1977
	300						Thompson et al, 1972
	500	640			20000		Pentreath, 1979
Ni						120-20000	Demayo, 1979
						1000000	Mudroch and Capobianco, 1979
	225-330			250-1000	640	11000-20000	Hutchinson, 1975
Pu	0.89-273			890-1800	276-1000	1050-100000	Amiard-Triquet, 1979

Table 2.1.2: Classification of radionuclides by biological role and periodic grouping. After Amiard and Amiard-Triquet (1980)

Radionuclide Class	Examples
I- No known physiological role	
a) Metals and Lathanides	Zr-95, Nb-95, Ru-103,106, Ce-144, Pb-210
b) Actinides	Th-230, U-234,238, Pu-237-244 incl., Np-237,239, Am-241, Cm-242,244
II- Chemically similar to elements with metabolic role	
a) Alkalai	Cs-134,137 (k)
b) Alkaline-earths	Sr-85,89,90, Po-210, Ra-226 (Ca)
III- Radioisotopes of metabolically important elements	
a) Transition metals	Mn-54,56, Fe-55,59, Co-57,58,60, Zn-65
b) Phosphorus	P-32,33
c) Hydrogen	H-3

elements from the littoral sediments through their root systems (Carignan & Kalff, 1980). The high phytoplankton BF values may be partly due to their high surface/volume ratios, and also perhaps to the cation-exchange properties of microfibrilar material often associated with algal cells (Leppard et al, 1977).

Perhaps the most important generalization to be made from Table 2.1.1 is that there seems to be no real trend toward increasing concentration at higher trophic levels, as has been observed in terrestrial systems. This may be partly explained by the efficiency of the excretion systems that higher aquatic animals have evolved (e.g., Hall, 1978).

2.1.5.7 Perspective on Interactions between Particulate and Soluble Phases of Water Body

Both biological processes (production, decomposition) and chemical processes (precipitation-dissolution, complexation, adsorption-desorption) control the distribution of an element between soluble and particulate phases of a water column. Whether biological or chemical processes are dominant in the control of distribution of radioisotopes in lakes between particulate and soluble phases is unclear. For some trace elements, adsorption isotherms (which imply chemical controls) have been found to be valid descriptors of the soluble-particulate relationship (e.g., adsorption of Cu onto δ-MnO_2) in certain concentration ranges; in other concentration ranges complexation has been found to be a valid description of the relation- ships (e.g., Cu with certain ligands, such as glycene; Vandenberg et al 1979). Use of mathematical models (e.g., Morel et al, 1975; Westall and Morel, 1978) present tools for resolving some of these relationships. Such question will continues to be areas for active research in the near future.

The removal of contaminants associated with particulates is dependent upon the settling rates of the different particulates. The settling rate depends significantly upon the particle's density. Organic particulates (specific gravity 1.02-1.1) settle much slower than sands (specific gravity 2.0-2.65).

Bioaccumulation factors are a fairly simple approach for examining soluble-particulate interactions. Their use requires many more measurements; use of functional groupings of elements could eliminate much duplication of research effort. To critically analyze available

data, standardization is required and aqueous concentrations should always be reported along with BF values. Data relevant to shield lakes are sparse and future research should attempt to supply them. Plants have by far the highest BF values, and concentration does not increase with trophic level.

2.2 LAKE SEDIMENT-WATER INTERACTIONS

In this section lake influences on physical and chemical properties of sediments and the measurement of the flux of water and material between the lake and sediments are examined. The currents of a lake are described above in 2.1.1 (lake physics), while the origins of the lake's energy for generating currents is described in 6.3 - atmospheric interactions with the lake. The net transport out of a tailings mass through a barrier and though the sediments under the influence of piezometric gradients of the lake groundwater system are examined in Section 4.0. General background material is available in Wetzel (1979), Baker (1980), and Goltermann (1977).

2.2.1 Lake Influences on Sediment Properties

Possible Measurements

1. Zones of accumulation and erosion
2. Water content of sediments

There has been a widely-held observation in limnology that there is a correlation between water column depth and the compactness of sediments. Only recently have these relationships been quantified. Such relationships provide information regarding the expected long-term characteristics of lake sediments if the mean depth of a lake is altered by disposal of substantial quantities of tailings in its sediments (i.e., if tailings volume is, say, one-half of the lake volume).

Hakanson (1980) has quantified the relationship between the sediment variables of bulk density, organic contents, grain size, degree of compaction and sediment water content, and between these variables and lake water depth. He suggests that sediment water content is a master variable which may be used to describe the others. For any one lake that he examined, the relationships between the water content of the top 1 cm of sediments and lake water depth are obtained - typically the sediments in shallow water have a lower water content and a wider variance about a mean value than sediments in deeper water. Based upon some arguments, he develops a parameter called the characteristic water content, which allows him to separate zones of lake sediment erosion from zones of accumulation. Then for a given lake, areas of erosion, transport, and accumulation may be determined using effective fetch and water depth as the master physical variables.

The chemical content of the sediments may be determined in a similar manner for those parameters which show a correlation with sediment variables; such correlations may not exist if industrial contamination has occurred. Based upon these correlations, he develops a sensitivity index which shows that different lakes and different basins have a different sensitivity to different toxic substances. Only a few of the metals considered (Cr, As, Zn, Pb, Cd, Cu, Hg) would have direct relevance to the lake disposal option for U mill tailings management; the approach is potentially useful for the long-term questions of tailings disposal because much of the thinking behind his analysis of sediment chemistry was directed toward assessing toxicological risks associated with sediments.

The physical data of Hakanson were developed from five soft-water lakes in Sweden with wide ranges in mean depth (6-40 m), surface area (20-6 000 km^2), and trophic status. Some care would be needed in the application of his numbers to Canadian lakes; and they would need to be checked with data from lakes near a potential disposal site. His method does provide a method for assessing the gross changes in sediment properties.

2.2.2 Net Removal Mechanisms of Trace Metals and Radioisotopes from Lake Water Columns

Possible Measurements

1. Amount of ionic precipitation
2. Amount of biogenic precipitation
3. Amount of complexed and adsorbed metals
4. Settling velocity of the different particulate fractions

The capacity of a lake ecosystem to absorb pollution associated with metallic and radioactive elements is largely a function of the degree to which these materials are removed from the water column and accumulate in the hypolimnetic sediments. Although net solution within and diffusion from the sediments into the water column may occur over short periods, the profundal sediments are generally viewed as a net depositional environment, or sink, eventually removing sedimented material from contact with the overlying water mass. The disappearance of dissolved matter from the water column (apart from lake outflow) is generally the result of one of three particulate formation and subsequent settling mechanisms: i) Ionic precipitation; ii) Biogenic precipitation; and, iii) Complexation of colloidal and particulate organic matter.

Such mechanisms are herein called removal mechanisms; their removal rate is equal to their particulate concentration times their settling velocity.

The section attempts to summarize our present capability to make predictions about these mechanisms and to identify areas of research that could enhance this capability.

Ionic Precipitation

The lack of sufficient knowledge about the speciation of most elements in fresh waters precludes definitive analyses of their solubility conditions. However, one can estimate solubility products for known inorganic salts which could be formed under a given set of conditions (Stumm and Morgan, 1970). For example, the conditional solubility product for a hypothetical metal carbonate MCO_3, or (MC), can be defined as:

$$pP_{(MC)} = \log K_{so(MC)} + 2pH + \log a_M + \log a_0, \qquad (2.2.1)$$

where pP is the negative logarithm of the calculated solubility product, K_{so} is the solubility product constant ($I < 0.2$; 25°C);

$$a_M = \frac{[M^x (aq)]}{[M_{tot}]} \quad \text{(x, considering the ionic species formed);}$$

and,

$$a_0 = \frac{[H_2CO_3]}{[CO_{3tot}]}$$

The actual concentration product can be measured directly in a lake or laboratory system. If the negative logarithm of the observed concentration product (pO) is less than pP a state of oversaturation is reached, and precipitation is a possibility. The amount of precipitation and its particle size, both of which must be determined for specific sets of natural conditions for each element. Hence, pO-pP may be viewed as a saturation index. Such indices are widely used in water distribution systems (e.g., the Langelier index, based upon saturation pH) and in watershed geochemical work (e.g., the calcite saturation index based upon the solubility products of calcium carbonate).

Baccini et al (1979) have determined pP and pO for zinc carbonate, zinc hydroxide, hydrozincite, cadmium carbonate and lead carbonate in

experimental limno-corrals (Gachter, 1979) at pH 9 and a total CO_3 concentration of 3×10^{-3} moles L^{-1}.

They predicted significant precipitation for only zinc hydroxide, and possibly a slight precipitation of zinc carbonate. Their results are perhaps applicable to other calcareous lake systems, but should under no circumstances be extrapolated to more dilute and less-buffered waters such as those of the Canadian Shield. From Equation (1), it is readily apparent that a decrease in pH will produce a decrease in pP (i.e., a logarithmic increase in the solubility product). However, pP will have a tendancy to increase in acidic waters because acidic waters tend to have higher proportions of H_2CO_3 in their total CO_3 pools; this causes the value of the last term of Equation (2.2.1) to increase. Since the effect of a decrease of pH will be larger than an associated increase in log a_o below pH of approximately 6.0 in many waters, pP will decrease for a given decreasing pH in such waters. Hence, these effects would dictate that pO would have to be much lower (i.e., the concentration product of the element much higher) before chemical precipitation could occur. The further lowering of the pH of shield lakes by acid deposition makes this possibility even more remote Recent. artificial acidification experiments in shield lakes (Schindler et al, 1980) showed that the observed solubility of most elements tested increases with acidity, although some elements (V, Hg) behave anomalously.

Another possible precipitation mechanism to consider is the adsorption of trace element ions onto inorganic particles such as $CaCO_3$. The scavenging of sedimenting calcium carbonate could be significant in calcareous waters, but probably not in shield lakes.

For the present, the lack of adequate thermodynamic data for predicting the speciation of many elements and the lack of wide application of present data to many lake systems hinders confidence in the use of solubility product calculations to predict chemical precipitation and subsequent settling. Research such as that currently in progress at the University of Waterloo, Ontario (J. Cherry, personal communication), at McMaster Universtiy (e.g., Vandenberg, 1979), at the California Institute of Technology (e.g., Morel et al, 1975) and at many other institutions have led to many useful predictions of speciation. Rate constants and particle sizes need to be determined for various types of lake chemistry and for functional groups of elements in order to complete the picture.

Biogenic Precipitation

The mineralization of dissolved material by aquatic organisms and the subsequent settling out of dead or senescent material is potentially an important component of contaminant sedimentation. Because of its high

capacity to concentrate trace elements, phytoplankton (Amiard-Triquet, 1979) and rooted plants (OWRC, 1971) must be considered as the primary agent of biogenic precipitation. Gachter and Mares (1979) concluded that zooplankton concentrate metals much less efficiently because of high excretion rates. It should be noted, however, that translocation of adsorbed elements to the carapace could be a sedimentation mechanism, expecially in organisms that undergo upwards of twenty moult cycles in a single year (Hall, 1978).

Macrophytes occur only in the littoral zones of lakes, and so are more significant in shallow lakes than in deep ones. In any case, macrophyte sedimentation occurs mostly into sediments that are in contact with the epilimnion; this deposition cannot be regarded as a permanent sink due to the potential for erosion to deeper lake areas, remobilization of metals and repeated uptake by macrophytes in subsequent years.

The rate of assimilation of trace elements by the phytoplankton is much faster than the rate of sedimentation. To estimate settling velocities of elements, one must first estimate the distribution of an element (M) between the biomass and the medium. The distribution coefficient D_M can be defined as:

$$D_M = \frac{[BM]}{[AM]} \qquad (m^3 \cdot kg^{-1}) \qquad (2.2.2)$$

where BM is the concentration of M in the biomass (mole/kg dry wt.) and AM is the aqueous concentration ($mole \cdot m^{-3}$).

The plankton biomass settling velocity can be expressed as the relation between the biomass present in the water column and that in the settled material. To estimate this rate (σ), Baccini et al (1979) used the formula:

$$\sigma = \frac{[\overline{SN}]}{[\overline{PN}]} \qquad (m \cdot d^{-1}) \qquad (2.2.3)$$

where $\overline{SN}$ is the mean settling flux of Kjeldahl nitrogen to the sediments ($mg \cdot m^{-2} \cdot d^{-1}$), and $\overline{PN}$ is the mean particulate Kjeldahl nitrogen concentration in the hypoliminion ($mg\ m^{-3}$). Because the distribution coefficient D_M is related to dry weight biomass, $\overline{PN}$ can be replaced by the total dry-weight biomass ($\overline{P}$, units kg dry wt m^{-3}) using the particulate nitrogen/dry weight ratio determined for lake plankton by Bundi (1979) of 90 ug PN/mg dry weight. The biogenic settling rate of the element in question ($\overline{S}_M$) can now be written as:

$$\overline{S}_M = [\overline{P}][AM] \times D_M \times \sigma \times F \qquad (moles \cdot d^{-1}), \qquad (2.2.4)$$

where F is the surface area of the stratified zone (m^2).

Baccini et al (1979) measured total settling fluxes for several elements and compared them with calculated values for S_M. The consistent correspondence between the two rates led them to conclude that formation of biogenic particulates was the major agent of sedimentation in their eutrophic lake ecosystem.

Examining Equation 2.2.4, it should not be assumed that an increase in aqueous metal concentration (AM) will lead to a directly proportional increase in S_M, because the capacity of phytoplankton to assimilate trace metals decreases with increasing concentration (Gachter and Mares 1979). This results in a reduction in the distribution co-efficient (D_M), which has the effect of lowering S_M. Thus, one must predict that an increased load of a dissolved element will lead to an increased final aqueous concentration of that element.

Another important point about Equation 2.2.4 is that S_M is a direct function of P, the total plankton biomass. Baccini et al (1979) worked in a eutrophic lake with a spring phosphorus concentration of 400 mg m^{-3}. Most Canadian Shield lakes have spring phosphorus levels on the order of 0.2-5% of that concentration, meaning that their planktonic biomass will be much lower. Although detailed experimentation on biogenic precipitation in shield waters is certainly warranted, one must assume that low productivity will severely limit biogenic particularization as a major sedimentation mechanism in such oligotrophic systems.

Adsorption by Humic Compounds

Because of their abundance in shield waters (Jackson, 1975) and their capacity for binding (Ramamorrthy and Jushner, 1975) and complexing (Mantoura et al, 1978) of trace elements, humic materials have been a subject of interest to researchers involved with heavy metal and radionuclide pollution problems. Szalay (1964) first proposed that humic acids be employed in the disposal of radioactive wastes. Humic matter is a collective term describing a group of complex, heterogeneous, brown or yellow acid polymers. It is a relatively stable by-product of the microbial transformation of a great variety of biochemical compounds derived from the dead cells and the extracellular products of various organisms, chiefly land plants and aquatic algae. Being relatively resistant to further decomposition, humic acids tend to accumulate in lake sediments. The humic matter present in a lake or in its muds is of two origins: allochthonous material, leached or eroded from soils and transported to lakes by surface runoff and groundwater flow, and autochthonous matter, formed from the cellular constituents and exudates of lake organisms. Humic matter in fresh water is largely

allochthonous, and its abundance is a function of watershed variables such as vegetation, climate and topography (Bordovskiy, 1965).

Humic substances are able to chelate or otherwise bind a wide range of ions. Carboxyl and phenolic groups are probably the most important ligands involved. This scavenging and subsequent sorption onto sedimenting particles has been postulated as the major settling mechanism for trace elements in shield waters (Payne and Pickering, 1975).

In attempting to predict the behaviour of trace elements in lakes, we are greatly limited by our lack of knowledge about element-ligand interactions (Jenne and Luoma, 1977). So numerous are the elements which could be considered as aquatic contaminants that an element-by-element research programme would take several decades. Recognizing the need for more immediate results, some researchers have attempted to classify metal elements into functional groups such that research or predictions on a given element could be applied to others of the same group. Nieboer and Richardson (1980) have used commonly available classification systems of chemistry to classify the metal or metalloid elements according to their affinity for different types of ligands. Rejecting the classification by specific gravity as relatively meaningless, they instead compared each element's covalent bonding index with its ionic bonding index. From this comparison they identified three groups: Class A (low covalent index, some with high ionic index); Class B (high covalent index, low ionic index); and, borderline (intermediate). Class A elements were found to seek oxygen-containing ligands, while Class B elements had an affinity for nitrogen- or sulphur-containing ligands. All the lanthanides, actinides, and alkalais, including radionuclide-yielding elements such as U, Th, Ra, Np, Pu, Am, Ce, Cs, Ac and Sr, fall into Class A. The fourth-row transition elements are in Class B, while fifth- and sixth-row transition elements tend to be borderline. Pb (high covalent and high ionic indices) and Cu (intermediate-to-high covalent index) were ambiguous, falling between Class B and borderline.

A different approach was taken by Jackson et al (1980), who investigated the affinity of 12 transition elements for particulate and colloidal matter as a function of fundamental physical properties. Surprisingly good regressions were found for crystal field stabilization energy (CFSE), electronegativity and ionic radius. The CFSE is an index of the potential strength of the bonds between transitional elements and their ligands; they found a direct relationship between CFSE and the rate at which those 12 elements disappear from the water column into the sediments. Electronegativity is a measure of ability to attract electrons from a donor; the direct relationship between the affinity of

these elements for both particulate and colloidal matter and their rate of accumulation in the sediments can be explained as a reflection of the degree and stability of covalent bonding. At least four factors could contribute to the explanation of the inverse relationship between ionic radius and affinity for adsorption sites: 1) the smaller the ionic radius, the higher the charge/radius ratio and thus the higher the electrostatic attraction to ligands, 2) an increase in cation polarizability with increasing size, 3) an increase in cation entropy with increasing radius; and, 4) steric effects in which large cations co-ordinate mainly with large, bulky ligands.

Of these three physical properties (CFSE, electronegativity, ionic radius), the relative importance of each in the behaviour of an element was dependent upon environmental factors such as the nature of the ligands and the ambient pH. For example, Hg was removed rapidly from solution at pH 6.7-6.8 due to its high electronegativity, but at pH 5.1 its large ionic radius slowed removal.

Given that the sedimentation rate of humic material can be adequately measured, the humic-linked sedimentation of any element could be estimated if the relationship between fundamental properties and ligand affinity were known for the group to which the element belonged. The predictive capacity for radionuclide behaviour in lakes would therefore be greatly enhanced if simple, functional relationships such as those found by Jackson et al (1980) could be developed for the alkali, rare earth and actinide groups.

The relationship between pH and the rate of trace element removal (Schindler et al, 1980) and between pH and remobilization from lake sediments (Bourg, 1979) is especially important in the context of acid precipitation impinging upon poorly-buffered shield waters. A possible explanation for these relationships is increased competition between protons and trace element cations for binding sites. In an effluent impact situation, competition by major cations such as Ca^{2+} and Mg^{2+} could also limit trace element adsorption (Mantoura et al, 1978).

2.2.3 Sediment-Water Transport

Possible Measurements

1. Present concentration gradients
2. Net vertical transport
3. Vertical particle movement (particle "hang-time")
4. Horizontal erosion
5. Biological grazing patterns

The main objective of studying sediment-water transport is to ascribe fluxes (upward or downward) to the significant transport mechanisms and to estimate net transport. Concentration gradients give an indication of the direction of transport of soluble species between pore water and lake water.

Net transport of a radionuclide out of the sediments is defined here as equal to the sum of vertical advection, transport of soluble species, vertical dispersive transport and sedimentation of particular matter. Advection arises from groundwater infuences and consolidation processes (sections 1.3.1, 4.0). Advective transport out of the sediment system is equal to the soluble concentration times the flow rate. If particles are moved, then this transport may be viewed as being due to erosive forces and is examined in section 6.4. If all particles eventually return to the sediments, the movement of contaminants associated with particles may be viewed in a net sense - it may be positive, zero, or negative depending upon whether uptake or release of the contaminant occurred while the particle was in the water column. Vertical dispersive transport occurs in the direction of a gradient (from a high to a low concentration) and is equal to the diffusivity coefficient (L^2/T) times the cross-sectional area of water (L^2) times the concentration gradient (M/L^4). Mass transport associated with sedimentation is equal to the settling velocity times the contaminant concentration associated with particles.

To measure these major fluxes, values inferred from mathematical models are typically used. Measurements can also be made of actual fluxes in the lake. For advection, use of a groundwater model gives one estimate. Models or laboratory measurements may be used to calculate diffusion; use of data for the pore water concentration of various substances acts as a check (e.g., DiToro and Connolly, 1980; Imboden, 1975). Methods for calculating diffusion are described in more detail in Section 1.3. The settling velocity may be measured by using traps (e.g., Burns and Hargraves), calculated from mass budgets (e.g., Dillon and Rigler, 1974), or inferred from the data of Hesslein (1978, 1979) (see Section 5.2) which results from a whole-lake spike of radionuclides.

Over the long term, it is expected that contaminant diffusion out of and sedimentation back into the sediments will be the two dominant transport mechanisms in most lakes. The calculation of diffusion appears to have excellent validity, particularly as the work of geochemists (e.g., Lerman and Taniguchi, 1972; Berner, 1974) has considered time scales of 100-1 000 years. The weak link is estimation of settling velocities, particularly of organic particles. From dating procedures and sediment models, a so-called sedimentation rate (0.01 mm - 1 cm/ yr.) is typically obtained. This is not the same as a sedimentation rate of organic matter from a lake water column. They can be related by appropriate considerations. This point can be quite confusing even to those who work daily in the limnology field. The term "settling velocity" and "sedimentation rate" are often used interchangeably and in a loose fashion. The key questions are:

1. whether one is referring to the water column or the sediments, and

2. whether one is referring to organic or inorganic particles in the water column.

In this work, we have attempted to maintain use of the term "settling velocity" to describe the water column and sedimentation rate to describe the sediments (the rate of burying of a given layer of sediment). In reference to the term "sedimentation rate", one further caution is in order - because sediment accumulates mainly in certain lake areas (accumulation zones versus erosion zones) while measurements are made at one or several sampling stations, care must be taken to ascertain whether data refers to one specific station, to an accumulation zone, or to the whole lake.

Bioturbation of sediments (caused by worms, clams, etc.) will increase the effective diffusion coefficient by an order of magnitude or more. Predictive tools are still sketchy but are evolving. For example, correlations are evolving between the diffusivity coefficient and the density of worms for describing sediment oxygen demand. Bioturbation will typically not influence any more than the surface 1-10 cm of a sediment with the larger thicknesses being for light sediments, and which are relatively decomposable. In thin sediments over tailings, effects of bioturbation will be substantial if the sediments are decomposable. Measurements of biodecomposability allow some inferences concerning the potential development of bioturbation. Methods for putting such measurements of biodecomposability of sediments into perspective are in the developmental stage.

Another method for calculating net vertical transport is the use of mathematical models. When one knows the water flows of the system (e.g., a river or lake), when one has an estimate of changes in

concentration over time, and when one is reasonably sure of the dominant biochemical and geochemical systems in the sediments, comparison of model predictions with observations in an iterative fashion until good agreement is obtained gives estimates of net fluxes. An illustrative example is the work of Onishi et al (1980) who looked as Cs-137, Co-60, Ru-106 and Sr-90 transport out of river sediments. One problem arises with this procedure if one has too many degrees of freedom (i.e., a large number of model parameters may permit the model to simulate the observed data with a variety of combinations of model parameters). For long-term, predictive purposes, problems arise if another independent data set is not available to check the calculated fluxes since one's confidence in the fluxes calculated increases with the number of data sets available for checking.

Perhaps the most fundamental problem facing any mining company with the potential use of the modelling methodologies, is the lack of any appropriate data. That is, one does not have data on a macro scale available for inferring micro-scale processes (e.g., settling, diffusivity). Then one must use data available from other sources for this purpose. For example, one might use data for adsorption and uptake onto biological particulates, and measured or calculated settling velocities for phosphorus nitrogen or biomass model to calculate a settling rate. This method is further explored in Section 5.2 (relationship of loading models to biological regimes). Such techniques have not been completely proven - they lie between being research tools and engineering tools. Their value lies in the development of a quantitative predictive model and in allowing a manager to develop "a feel" for what the possible size of errors are. Such techniques are illustrated by radionuclide transport calculations for river systems by BEAK (1980b). The methods of BEAK (1980) are relatively coarse for present requirements of the lake disposal question.

Horizontal erosion represents the movement of material from one area to another under the influences of currents. Detailed hydrodynamic models have the ability to provide fairly good details on erosion in rivers, but have not been developed as adequately for lakes. Work similar to that of Hakanson may be sufficient for lake disposal questions if one compares net vertical transport. Comparison of net vertical transport with sediment erosion represents a potentially useful method for looking at the transport of contaminants out of a sediment. Certain groundwater flow rates would be sufficient to move sediments (e.g., hydrothermal activity in mountain and other seismically active areas). Wind-driven currents and earthquakes could cause horizontal erosion. A key question for each type of movement is whether the particle returns to the lake sediments and what are the nature of its interactions with lake water while it is suspended (this assumes that all eroded particles which do not leave the lake system

through its outflow will eventually settle). The net movement of contaminants from the sediments due to particle erosion may be positive, zero, or negative depending on the uptake release reactions in the water column. Partial answers concerning the magnitude of this contaminant movement (e.g., tailings leachate movement) may be obtained by examining studies related to sediment dredging (e.g., Lee et al, 1978), but the total answer will depend upon time the particle remains in the water column, i.e., the particle "hang-time".

Biological grazing of sediments represents a means of transport from sediment-organic constituents up the food chain (bacteria, benthos, bottom-feeding fish). Knowledge of present patterns (areas and densities of benthos in the sediments, fish feeding habits, bioaccumulation and bioconcentration rates of different fish) and use of dosimetry calculations similar to those of BEAK (1980) for terrestrial systems, but applied to a lake food web, permit assessment of this transport.

2.2.4 Perspective on Lake Sediment-Water Interactions

For lake disposal questions of U-mill tailings, sediment-water interactions are the last critical step for controlling the ratio of movement of tailings leachate into lake water at an acceptable rate. Much is known about these interactions much is also unknown.

Because lakes tend to be net depositional areas, net transport over geological time scales of many materials tends to be towards the sediments. With the placement of tailings in the lake, there is a potential for a temporary perturbation of such a relationship.

Long-term release from sediments to the water column is expected to primarily be by soluble transport (diffusivity, advection); return will be by settling of contaminants associated with particles. This potential for return then gives a parameter which may be called "net release" ("NR"). If net release is greater than zero, then a useful management parameter is:

$$\frac{NR}{W}$$

where W is the input ratio to the lake (see Section 5.0 for its determination). Here NR and W each have units of mass/time. BEAK proposes that this parameter may be used for determining acceptable bleed rate from the sediments.

Chemical precipitation and biogenic precipitation can be important agents of trace element sedimentation, particularly in calcareous

waters; the behaviour of radionuclides in these systems needs further immediate investigation. For Canadian Shield Lakes, humic matter scavenging is probably more significant, so radionuclide behaviour should be studied in this context and also with respect to acidification. If the necessary research were to be carried out, it should be possible in the foreseeable future to make reliable predictions about pelagic removal rates and sediment retention capacities for most trace elements in Canadian lakes.

Occasional perturbations (e.g., hydrothermal springs, wind driven currents, earthquakes) may erode some sediments. Knowledge for predicting the extent of contaminant release is less developed than for predicting net release. Use of elutriate tests in model systems together with calculated estimates of hang-time (the time that an eroded particle is in suspension) may provide a method for scoping the magnitude of this transport method compared to that of net release.

3.0 HYDROGEOLOGY

Major concerns within hydrogeology are: i) the stability of the bedrock acting in its capacity as a foundation for the lake; ii) the presence of significant groundwater aquifers which connect to surface waters; iii) the magnitude of groundwater transport into and from the tailings area; iv) the physical and chemical stability of aquifers; and v) the ability of porous media and bedrock to adsorb any contaminants migrating from the tailings areas. These latter points generally describe the rate and duration of transport of contaminants through the bedrock and the ability of the bedrock to attenuate such transport. Geological criteria have been extensively development for disposal of high specific activity wastes (e.g., Gray et al, 1976). Such criteria consider both transport and structural stability of bedrock; our approach tends to emphasize transport.

3.1 PHYSICAL STABILITY OF BEDROCK

Possible Measurements

1. type of bedrock
2. presence of major faults
3. history of seismic activity
4. historical uplift character of plate
5. fractured nature of bedrock
6. long-term maintenance of fracture characteristics

The physical stability of bedrock provides an assessment of the long-term potential for disturbance of the watershed or tailings due to plate uplife or due to extrema such as earthquakes. If significant plate uplift occurs, the depth of water overlying the tailings may decrease. If seismic activity is probable, the tailings may be mobilized or the fracture character of the bedrock increased leading to increased groundwater transport. Hence, by way of analogy, the bedrock may be viewed as a foundation for the tailings-lake system. Bedrock stability is critical for maintaining the long-term physical stability of the tailings. Typical engineering approaches to the analysis of foundation problems are given by Holtz and Kovacs (1981) and Goodman (1980).

The type of bedrock (sedimentary, metamorphic or igneous) allows overview assessments about long-term geological and geotechnical stability and long-term transport characteristics of groundwater. Further subdivision according to mineralogy and geological history is also useful. For example, sedimentary rock such as limestone is much more prone to long-term dissolution than igneous rocks.

The presence of major fault lines and their characteristics provide a qualitative guide for assessing the overall stability of the bedrock during recent geological history. Seismic activity records may be used as a measure of recent stability. The development of earthquake zones for building codes and foundation engineering purposes provide guides to anticipated behaviour. Quantitative seismic records have only been maintained for a short period of time. Human historical records (e.g., observations on the frequency of shaking of china plates in a farmhouse) are only quite recent; qualitative observations by local people form a longer history, but frequently have not been committed to paper. Oral histories (taken by historians, sociologists, etc.) are helpful in this context.

Many areas of Canada (e.g., Hudson's Bay, St. Lawrence River) are emergent zones due to isostatic rebound after the recent (geologically speaking) retreat of glaciers. The emergent character is also due to larger scale tectonic processes. The rates are of the order of a few cm/century. Over a 10 000 year time frame, this behaviour will have significant effects upon erosional characteristics of a watershed. It probably will not have a significant effect upon "foundation properties" of the bedrock, unless the plate itself is prone to earth tremors or other seismic activity.

The fractured nature of the bedrock is a measure of the potential for groundwater transport and of the potential for long-term change of fracture characteristics. Actual groundwater transport will occur only if the fractures are connected. A high degree of fracture indicates a potential for increasing fracture if the right stresses occur.

The long-term maintenance of fracture characteristics is mainly a function of future stresses imposed upon the rock. Consider three examples. One example is the effect of ground motions such as earthquakes upon bedrock. If a sudden horizontal shear force is imposed upon the bottom of a geological formation, the inertia of the formation will attempt to resist the motion. If the motion cannot be translated perfectly and in an unattenuated fashion, there will be a differential shear force imposed upon the bedrock. If the tensile strength of the bedrock is insufficient to resist the differential shear force, the bedrock will fracture. Knowledge of the tensile strengths of bedrock permit some inferences concerning the probability of fracture as a function of the magnitude of the seismic events. Knowledge from the field of Rock Mechanics permits good assessment of the fracture potential of bedrock, if the actual tensile and compressive strengths of the various geological strata have been measured. Unfortunately, the future fracture character caused by earth movement cannot routinely be estimated in a precise deterministic fashion. This is partially due

to inadequate geological mapping but is more due to our present inability to predict in a sufficiently precise manner seismic activity and the associated stresses.

A second example is the exposure of a groundwater-bearing stratum to the atmosphere by events such as on hillsides. The depth of penetration of frost is a measure of the potential zone of bedrock whose fracture characteristics will change over time. If these zones are small and sufficiently removed from transport zones which connect to tailings areas, these effects will be negligible. However, it will be significant in downstream areas if frost-related fracturing, weathering and erosion either i) remove sufficient bedrock over 10 000 years so that groundwater transport through the tailings area are significantly increased or ii) reduce the stability of the sill controlling the level of the lake.

A third example is where a significant change in hydraulic pressure changes the fracture character of a bedrock. Consider a bedrock underlying a lake. At any point in the bedrock, its fracture character is such that it is at equilibrium with existing stresses. Consider three possible cases. If a reservoir is built one to two kilometres away, upon a stratum which connects the reservoir waters with the lake selected for tailings disposal, the hydraulic head of water caused by the reservoir could increase the fracture character of the whole stratum. If a landslide blocks a river valley, it may create an impoundment upon a stratum which is connected to a lake and this hydraulic pressure again will alter the fracture character of the strata. A third case is the replacement of substantial volumes of lake water (e.g., 50%) by tailings; such additional pressures will change the fracture characteristics of underlying bedrock. All examples of fracturing may occur immediately or at some future point in time.

The key to preliminary analysis of these problems is to examine pressures. In all three examples, the driving force for potential fracture is the change in pressure. In the lake or reservoir examples, previous stresses were unbalanced by the extra head of water translated horizontally through the geological stratum. In the tailings example, the forcing pressure is the added overburden pressure and may be approximated as being equal to the specific gravity of the saturated tailings minus 1.0, times the height of the tailings.

3.2 CHEMICAL STABILITY OF BEDROCK

Possible Measurements

1. dissolution characteristics of bedrock for different aggressive waters
2. potential precipitates from groundwater constituents

One major difference between groundwaters of Southern Ontario and many other areas of Canada, and those of the Precambrian Shield is their differences in concentrations of carbonate and bicarbonate. This occurs partly because southern Ontario is underlain by carbonates which are more amenable to dissolution than are the carbonate-poor igneous rocks of the Precambrian Shield. If the groundwater has a relatively neutral pH, it has a much lower ability to dissolve carbonate minerals than if it is acidic.

One possible measure of dissolution-precipitation characteristics is a dissolution potential. Such measures are typically formulated in terms of indices which give the potential for dissolution based upon solubility of specific minerals and actual characteristics of the groundwater. For example, the groundwater in the CRNL disposal areas at Chalk River appears to be at equilibrium with calcite (Jackson and Inch, 1980). Such indices have been developed for surface waters and groundwaters, although similar indices occasionally bear different names depending upon the field of expertise in which they were developed.

For calcite, an index such as the Langelier Saturation Index (LI) may be used. It is equal to the actual pH minus the saturation pH (with respect to $CaCO_3$). If the LI is negative, precipitation may occur, if the LI is positive, dissolution may occur. This index is widely used in the water treatment industry (e.g., municipal water supplies) and in the electrical power industry (e.g., cooling water systems). It requires modification only for site specific variations (see Kemp, 1971; Langelier, 1946; McGaughey and Matson, 1980). There is one caution in its use. It denotes a directional tendency - that is, it describes the potential for dissolution but it does not make statements about the rate of dissolution. The rate of dissolution is a function of the potential for dissolution as well as other chemical properties of the groundwater. An alternative index is the calcite saturation index (CSI) developed by J.R. Kramer (personal communication, McMaster University). It is equal to the difference of the log of the product concentration of calcium and carbonate and the log of its solubility product and has been applied to various surface and groundwaters. It essentially assesses the same precipitation potential as the LI but has a different numerical scale.

For groundwater, the indices have received systematic treatment and use for many minerals. The saturation index (SI), is equal to the product of reacting species forming a mineral divided the solubility product (Troup, 1974). Indices are available for a host of minerals including calcite, dolomite, gypsum, etc. Sometimes it is formulated on a molar concentration basis, but often on a log of the molar concentration basis (Freeze and Cherry, 1979). In the latter form, the CSI of Kramer corresponds exactly. If greater than 1, the mineral will precipitate; if less than 1, it will dissolve. The concept of the SI is the basis of many geochemical equilibrium models (e.g., WATEF) which predict what minerals a given groundwater is at equilibrium with.

If the groundwater is supersaturated with respect to certain minerals (e.g., iron, sulfide) or if specific exchange reactions can occur between the groundwater and bedrock, chemical precipitation or adsorption reactions may occur to decrease the size of fractures.

3.3 GROUNDWATER FLOW CHARACTERISTICS

Possible Measurements

1. porous media versus fracture media characteristics
2. directional permeabilities
3. bulk diffusivity coefficients
4. fingerprint features

The examination of directional permeabilities, anisotropic dispersion characteristics and fingerprint variables should be made with two particular questions in mind. They are: Does the groundwater flow have the potential for causing significant upflow velocities through the tailings? Does the groundwater flow have the potential for introducing significant contaminant transport to downstream receiving water (i.e., the surface ecosystem) due to leaching from the tailings?

The quantification of groundwater flow typically describes transport in terms of one of two processes: i) an advective (also called convective) process described as a velocity (e.g., m/day) or flow rate (e.g., m^3/day) between two points, and ii) a diffusive or dispersive process (e.g., m^2/day). Advective flow occurs due to a hydraulic or pressure gradient. Diffusive flow can result both from molecular and eddy processes while dispersive flow can result from molecular, eddy, and advective processes. Whereas eddy processes are dominant in lakes, molecular processes are the dominant form of diffusion in groundwater. Hence, diffusion is typically described as being a molecular process whereas dispersion is mostly strongly a function of advection in groundwaters.

On a microscale, groundwater trasport is controlled by various hydrogeologic properties of the medium through which it is moving. They are reviewed in some detail by Kilborn (1980). These properties are referred to as "porous media" characteristics by Kilborn. This is somewhat of a misnomer. Much of the original work on quantification of groundwater flow comes from quantification of flow through porous media. This work starts from describing one-dimensional flow (advection, dispersion) through beds of ideal spheres packed in various manners (e.g., the packed bed reactors of water treatment plants. The addition of anisotropic character is a second step. The Laplace equations for flow in two directions is an example of isotropic flow in two directions (e.g., Terzaghi and Peck, 1968; Cedergren, 1977). From this basis, attempts have been made to describe flow through fractured media. Due to the extreme difficulty of relating fracture characteristics (a 3-dimensional problem in the local sense) on a microscale to macroscale properties which are more amenable to engineering description, typical mathematical concepts attempt to characterize fracture media flow by the use of analogy to porous media flow. Such relationships render quantification of fracture flow more tractable. Because fracture flow often occurs in one particular direction (due to the direction and interlocking character of the fractures), resistance to flow through bedrock is often described by directional permeabilities and directional diffusivity coefficients.

Additional complication for describing contaminant migration arises from the temporary migration of contaminants into "dead" space and the adsorption onto the support media. For modelling water flows, description of flow often considers only advection. However, there can be large volumes of water in bedrock or in granular materials which are not in the flow path, which are only marginally connected to flowing water, but into which contaminants can migrate. Use of short-term data for calibrating groundwater flow models and for describing solute transport may neglect such zones and slow transport processes to them. Over the long term, slow transport processes may be very significant. This "dead" space (or temporary storage) and adsorption are two phenomena with potential for significantly changing long-term estimates of transport behaviour when very short-term field data are used for mathematical model building.

To adequately determine these relationships, it is suggested that use be made of modelling studies such as that of Bobba and Bukata (1980). They constructed a model for groundwater transport through homogeneous and anisotropic, unconfined aquifers. They present nomographs for the effect of hydrogeological properties of advection, dispersion and chemical adsorption upon the buildup, steady-state, and recovery phases of subsurface contaminant transport. Because radionuclides such as Ra and

U follow a linear adsorption behaviour over the concentration range typical of groundwaters, these types of nomographs permit checking on the transport characteristics of radionuclides through hydrogeological regimes (porous media, fractured media). The authors find that of principal velocity ratios, dispersivity ratios and adsorption coefficients, the principal velocity ratio (i.e., the degree of aquifer anisotropy) is the dominant variable affecting transport of subsurface contaminants over the short-term (90 days). For geological time scales, these conclusions may not be applicable because a long-term time-scale is of concern. However, development of such nomographs for the long term, in a fashion similar to those of Bobba and Bukata, is the tool required for environmental decision making. Use of the field studies such as those of Jackson and Inch (1980) provide specific data for such nomograph development.

Fingerprint features are those which key to abnormal geological features. Such features may greatly increase groundwater flow or retard contaminant migration. Such features include glacial river channels presently covered by overburden, bedrock channels such as evidenced by caves, and evidence of hydrothermal activity.

3.4 BIOCHEMISTRY OF GROUNDWATER

Possible Measurements

1. organic characteristics of recharge areas
2. potential for oxygen migration

The potential for leaching action and mobilization of radionuclides by groundwater is controlled in part by the groundwater pH, the groundwater Eh and the biochemical systems controlling these variables. A more detailed description of the basic concepts, biochemical systems and geochemical systems is summarized in section 4.0. Descriptions of several scenarios are given by various authors such as Jackson and Inch, (1980) and Champ et al, (1979).

The leaching potential or mobilization potential for radionuclides, heavy metals, sulfides and other materials is strongly controlled by the pH and Eh of the groundwater. Acidic conditions and oxidizing (high Eh) conditions aid in mobilization of certain metals; acid conditions and reducing conditions (low Eh) cause mobilization of others. Unless material such as sulfide is directly leached from bedrock (a geochemical mechanism), it is a reasonable hypothesis that a dominant control on the degree of oxidation-reduction in groundwater is bacterial action. Geochemical mechanisms such as formation of minerals act to

control the actual chemical species involved and the precipitates formed and to mediate the Eh of the system. Both biochemical processes (bacterial processes producing H^+) and geochemical mechanisms (e.g., dissolution of carbonate rocks) strongly influence the pH. The contribution of biochemical processes may be assessed by examining the efficiency of H^+ production per mole of dissolved organic matter consumed, while the contribution of geochemical processes can be assessed by looking for sharp breaks in geology and geochemistry (e.g., redox gradients, changes from igneous rock to sedimentary; changes from groundwater dominated by carbonates and nitrates to groundwater dominated by sulfate - the latter situation is particularly found in the Prairies or in oceanic areas in Canada).

For bacterial activity to occur in aerobic or anaerobic regimes, an energy source is needed. The common energy sources are particulate organics (usually non-transportable) and dissolved organics (easily transportable). These organics have varying degrees of biodecomposability (amount of energy available for bacterial growth and hence electron transport per mole of carbon). Carbohydrates are more degradable than are organics leached from tree leaves and bark (e.g., humic acids, fulvic acids).

If groundwater originates in nearby marsh areas, there is a good probability that it will have reducing tendencies. If the groundwater aquifer is near the surface, oxygen may penetrate to a sufficient depth to prevent anaerobic conditions. If the aquifer is confined, it will attain reducing conditions if isolation from sources of free oxygen is prolonged.

For mathematical treatment of this problem, a key variable is the rate of consumption of organics in the water compared to the rate of transport of oxygen. Such a variable has not evolved for practical application to the problem at hand; however, we assert that it is readily developable, using concepts available from the literature relating to chemical and pollution control engineering, cooling water technology, and lake sediment oxygen demand.

This variable would permit statements concerning the potential reducing characteristics of groundwater (and also sediment pore water or tailing pore water) and the potential mobilization character of groundwater as a function of its history of travel.

Perspective on Hydrogeology

Hydrogeology has three particularly relevant aspects to lake disposal of tailings. Firstly, the "foundation" or geotechnical stability of the bedrock allows one to assess the potential for disturbance of the tailings due to extrema such as seismic activity. Secondly, the direction of groundwater flow and its flow rate give an indication of whether there is potential for significant vertical flow and for contaminant transport through the tailings. Thirdly, the extent of fractures in bedrock, the voids volume in overburden and the degree of connection of these fractures and voids allow an assessment of the potential for formation of a groundwater contaminant plume. The actual size of the plume will decrease if significant chemical precipitation and adsorption occur. Plume size is a critical parameter for assessing whether or not the plume would allow significant discharge of contaminants to surface waters. How the chemical and physical characteristics of the parent media change over the long term is critical.

4.0 INTERACTIONS BETWEEN TAILINGS, LAKE SEDIMENTS AND HYDROGEOLOGY

Possible Measurements

1. direction of advective flow
2. fingerprint features for advection
3. advective and diffusive transport direction
4. flux of chemical elements into tailings
5. flux of chemical elements out of tailings
6. probable long-term geochemical and biochemical mechanisms
7. detention of materials in sediment-tailings systems
8. susceptibility of receiving waters to these fluxes

If contaminants migrate out of the tailings, there are two main directions of concern: upward through the sediments into the lake and downward into a groundwater aquifer which is connected to surface waters at some downstream location.

Philosophically, it would appear to be beneficial to encourage transport into a deep groundwater (e.g., 200 m), provided it does not interface with surface waters. This statement is made from the premise that due to solid state diffusion through tailings, it is probable that a significant fraction will diffuse out of the solid into pore water over a period of time equal to or greater than 10 000 years. Biochemical processes will accelerate this movement. The question is in which direction should migration be encouraged - into groundwater or into surface waters?

The assimilative capacity of bedrock is a useful concept, provided that the groundwater will not be a water supply source (e.g., wells) in the foreseeable future and provided that it does not intersect with nearby surface waters. The assimilative capacity of surface waters is unclear. If background in a watershed (due to weathering processes, and due to atmospheric interactions with the drainage basin and water table, see section 6.4) is substantial (e.g., 5 pCi/L for Ra) and the bleed rate out of tailings is low (e.g., resulting in 0.5 pCi Ra/L), the question arises as to whether, with the high natural background, there is any assimilative capacity. Alternatively, if one could direct the equivalency of 0.5 pCi Ra/L into groundwater and even if it does intersect with downstream waters, it may be transported through bedrock with a sufficiently high adsorptive capacity that it enters the surface water with an equivalent concentration of 0.1 pCi/L. If background groundwater concentration is 5 pCi/L, it is conceivable that transport directed towards groundwater still may be desired because of the ad-

sorptivity for the radionuclide by the bedrock and a smaller incremental radioisotope concentration entering the surface water (this assumes that both surface water and groundwater have natural background concentrations of 5 pCi/L, due to leaching of radioisotope material from exposed bedrock for surface water and internal fractures for groundwater).

In a relatively pristine environment, some debate will ensue concerning the concept of an "assimilative capacity" of the environment for radioisotopes. If, for example, migration is encouraged into deep bedrock areas (200 m), the radioisotopes may be dispersed at an acceptably low bleed rate to give a very low level of available contaminants. An obvious social fear arises from the potential use of such groundwater in perhaps 1 000 years. This question will arise, inevitably, because of the present realization that many aquifers which are water supplies for many muncipialties (particularly in the U.S.) are now contaminated by leaching of toxic chemicals from old landfills and chemical dumps. This environmental concern will increase over the next few decades as more analyses and monitoring are carried out, and as the front of the migrating leachates (a few cm to a few hundreds of meters per year) reaches more public and private wells.

The answer to this question has to be one of balance: is the dispersed concentration sufficiently low? Which alternative is better? — dispersal to surface waters or to groundwaters? Is disposal in a depositional environment or in an erosional environment a better alternative? Disposal in a depositional environment (e.g., a lake) for a time-scale of 10 000 years has a much higher probability of being stable than deposition on an erosional environment (e.g., the side of a hill, a lake's drainage basin). Disposal of solid wastes has been conducted in marshes (a depositional environment); unfortunately, many marshes are recharge areas for surficial groundwater aquifers resulting in the appearance of mobilizable toxic chemicals in the surface biosphere within a few years to a few decades. Hence, the depositional environment should be one in which groundwater transport is localized and deep. Whether dispersal to surface water or to groundwater is the better alternative will be site dependent. These examples suggest a few parameters for deciding this question: i) the probability of future use of the groundwater, ii) the relative increment above background concentration in surface water and ground water, iii) the discharge of groundwater to areas with downstream waters and the assimilative capacity afforded by such uptake mechanisms as adsorption.

In the remainder of this section, the following reasonable assumptions are made. First, all significant transport from the tailings will be

as soluble leachates moving through pore waters. Second, soluble transport will occur due to the advective transport (dominated by the regional groundwater-lake flow system and augmented by squeezing of pore water out of the tailings-sediment system through consolidation) and by diffusion. The relative dominance will be site specifica, but vertical transport through deep lake sediments is typically dominated by diffusive processes. Third, some transport will occur if lake-bottom currents are sufficiently strong to erode the sediments; the significance of this transport will depend upon contaminant concentration on the eroded particulates and the rates of uptake and release while the particles are suspended. Fourth, geotechnical extrema such as an earthquake will normally cause the tailings-sediment system to raise and lower in a fairly intact fashion. No slumping due to gravitational effects would be expected if the tailings and sediment are fairly level. Sediment cohesion is a useful parameter for assessing the question of slumping. Some transport may occur (vertically downward) if a fault opens up under the tailings.

A first requisite for assessing the transport of soluble materials is knowledge of the hydraulic gradient of the watershed's groundwater fields and of the regional groundwater fields. One can obtain some of this information by assessment of the piezometric (hydraulic) gradients and the directional permeabilities (vertical, horizontal) of the different bedrock strata. However, insertion of a tailings system, and a liner which has a permeabiity substantially lower than that of a fractured bedrock, may significantly change the watershed contained groundwater flow field and even the regional groundwater flow field, depending upon the depth of the lake.

Fingerprint features of the watershed-lake system aids in assessment of potential nuances in the advective flow field. Fingerprint features of special value include the potential for fractured bedrock found by inspection of outcrops, regional hydrothermal activity, and regional artesian fields.

The direction of advective flow defines the direction for transport of a contaminant due to groundwater flow and the probable direction for net transport of a contaminant. Diffusive transport occurs in the direction of a concentration gradient; hence, it may occur both in the direction of and in the opposite direction to groundwater flow. If advection is sufficiently low, or if the concentration gradient is sufficiently high, diffusive transport will be greater than advective transport. For many lakes, vertical transport out of the deep sediments is governed by diffusive transport because the hydraulic gradient is essentially horizontal and equal to the height of lake water or

directed downward; only at the lake edges where groundwater gradients are substantial is vertical advection consistently important. The flux of chemical elements into tailings gives a measure of the driving force for changing probable geochemical and biochemical systems controlling uptake-dissolution mechanisms in the tailings (e.g., an acidic inflow, a carbonate buffered inflow, an inflow high in organics).

The flux of a chemical element out of a tailings through sediments into the lake water or into the groundwater defines the bleed rates and the direction out of the sediment-tailings system. This is the parameter to be used by managers to determine whether or not transport rates are acceptable.

The flux of chemical elements into and out of the tailings is a measure of the long-term rate of breakdown of existing biochemical and geochemical mechanisms within the tailings which control the uptake-dissolution mechanisms. The mass in the tailings divided by the net mass flow out of the tailings (if the net flux is outward) gives a measure (units:yr.) of the rate of removal of the material. This ratio is a measure of the detention time of the material in the tailings. If the detention time for several materials which control a geochemical mechanism is relatively short compared to 10 000 years, then there is a good probability that the geochemical mechanism will change. For example, if the detention time for calcium carbonate is 800 years, it means that at some time period longer than 800 years there is a probability that all calcium carbonate will be dissolved. If this occurs, a candidate system for controlling pH may be the H_2S-S^{2-} system. However, whether such a shift occurs will depend upon the relative mass of CO_2-CO_3^{2-} transported into the tailings-sediment system compared to the dissolution rate of various elements associated with other candidate controlling systems. The quantity of CO_2-CO_3^{2-} may still be sufficient to control pH, with some shift from the pH of the inflowing water occurring because of dissolution of sulfide and accessory minerals. If this occurs, then the pH is controlled both by transport and by geochemical mechanisms.

Over the long term, geochemical regimes will be controlled by the mass of different materials initially present in the tailings-sediment system, by a mass balance in the pore water governed by transport into the tailings-sediment system, by mass transport out, and by the relative rates of uptake and dissolution reactions occurring between soluble and particulate phases. Biochemical regimes will be controlled by similar factors, with the different microorganisms which develop changing the relative rates of uptake and dissolution reactions.

To provide some perspective to the question of relative importance of biochemical and geochemical mechanisms, consider the following examples. All examples describe tailings in a lake sediment with different possible combinations of biochemical reactions. In all cases, oxygen diffuses from the overlying water into the upper sediments layers (typically a few mm). In the aerobic sediment layers, bacteria use organic matter to form carbon dioxide. As well, reduced species (e.g., S^{-2}) diffuse from deeper anaerobic layers and are oxidized. In deeper sediment layers anaerobic reactions occur and methane is formed either from CO_2 or from organics.

In case I (Figure 4.1(a)), sulfide and radionuclides are leached from tailings and are transported to groundwater (advection, diffusion) and to lake water (diffusion); the Eh will be controlled by sulfide evolution (a geochemical mechanism). In case II (Figure 4.1(b)), similar reactions occur as in case I, except that organics in the tailings (either there initially, or transported by groundwater) allow the development of anaerobic bacteria. Since the Eh associated with this case may be lower than that associated with the leaching of sulfide, the biochemical mechanism will control Eh. In case III (Figure 4.1(c)) similar reactions to case I occur, except that high nitrate and sulfate concentrations exist in the lake water causing the development of an anoxic layer (nitrate reduction) and a sulfate reduction layer. In case III, it is assumed that sulfate reduction is greater than the sulfide leaching rate and that bacterially-induced sulfide production controls the Eh of the tailings. Such lakes as in case III could include brackish water such as are typical of southern Saskatchewan and old sea water intrusions presently cut off from the sea. In case IV (Figure 4.1(d)), a similar set of reactions as in case I occurs, except that there is sufficient reduced iron present (Fe^{2+}) to precipitate the sulfide (perhaps a vivianite). Case IV raises the possibility of the coprecipitation of a radionuclide with vivianite which is mechanism for reducing the bleed rate of a radionuclide from the tailings mass.

In Table 4.1, an attempt is made to place the Eh question into perspective. In the centre of the table, the typical biochemical systems found in lake water and its sediments are summarized and their relative degree of oxidizing or reducing conditions. In aerobic lake water, reduced substances are oxidized - Mn^{2+} and Fe^{2+} oxidation is mainly chemical, while ammonia and organic matter (CH_2O) oxidation is mainly biological. Sulfide oxidation can occur both chemically and biologically. The relative order from Mn^{2+} to CH_2O gives an indication of the degree of oxidizing conditions theoretically required for these oxidations to occur.

Figure 4.1(a): Case I: Sulfide leached from tailings, no organics in tailings (RN = radionuclide)

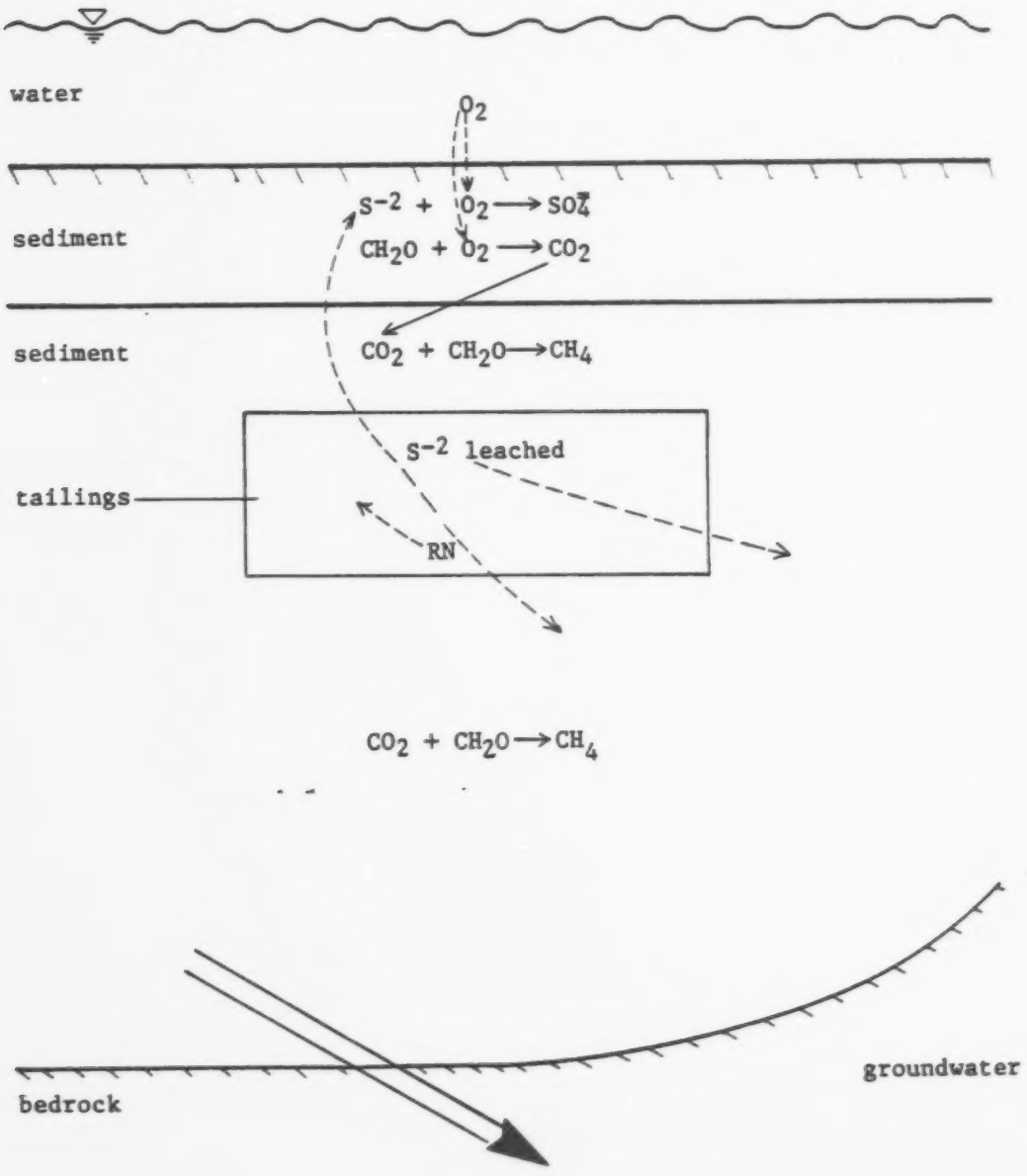


Figure 4.1(b): Case II: Sulfide leached from tailings, organics in tailings (RN = radionuclide)

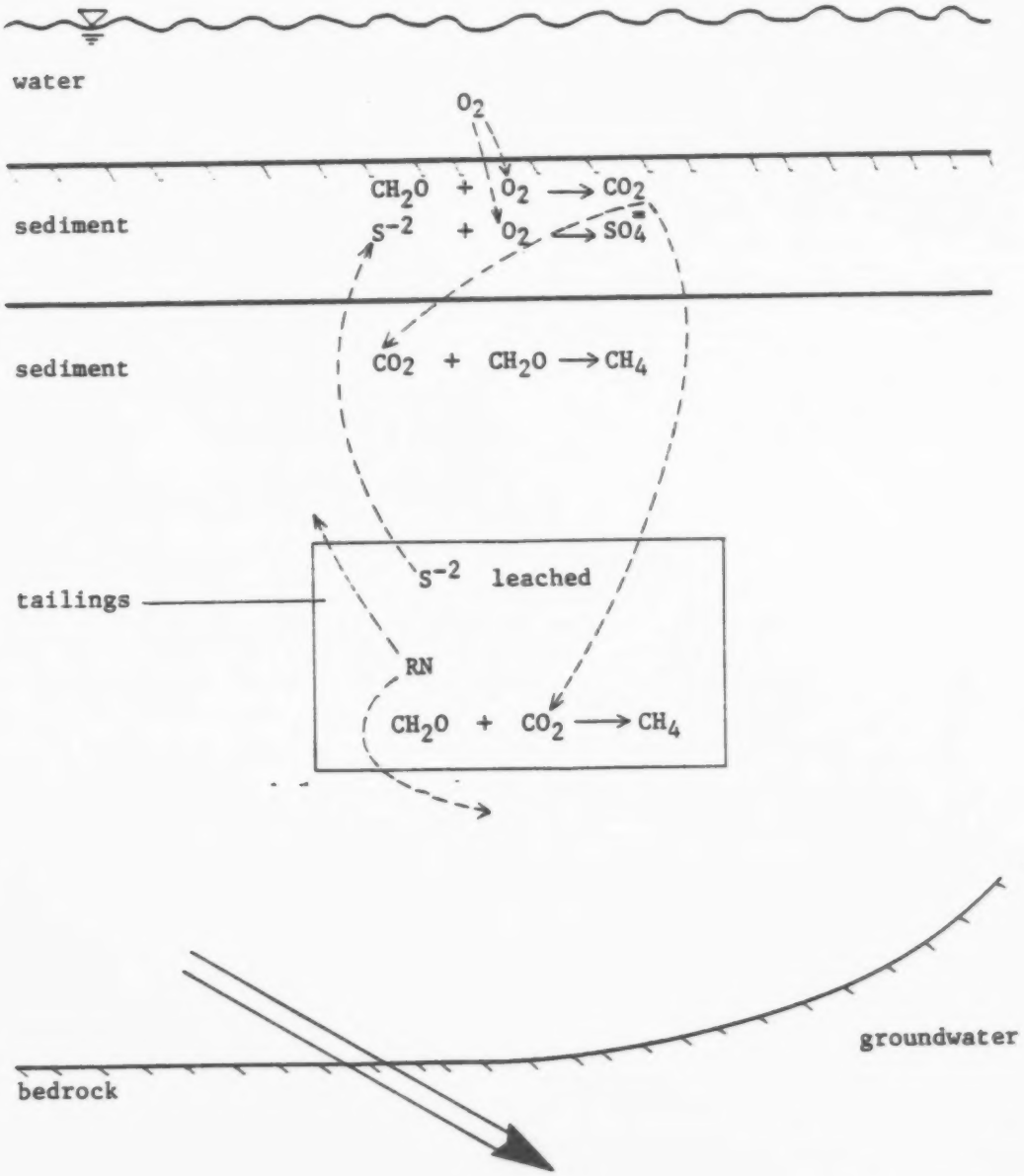
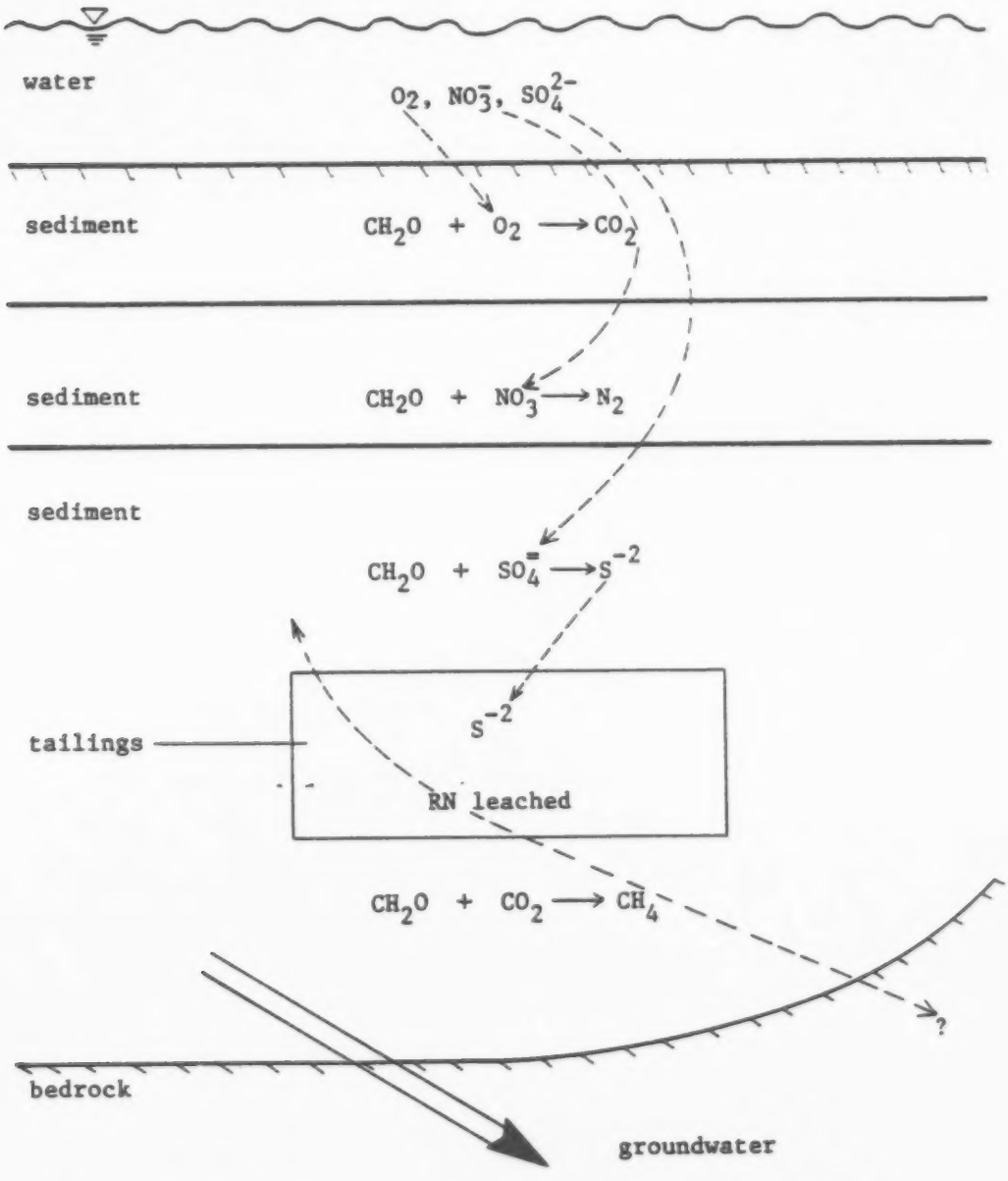


Figure 4.1(c): Case III: No sulfide leached from tailings but sulfate and nitrate in overlying water (RN = radionuclide)



Comments: no organics in tailing preclude bacterial life

Figure 4.1(d): Case IV: Sulfide leached, organics in tailings and iron minerals are present (RN = radionuclide)

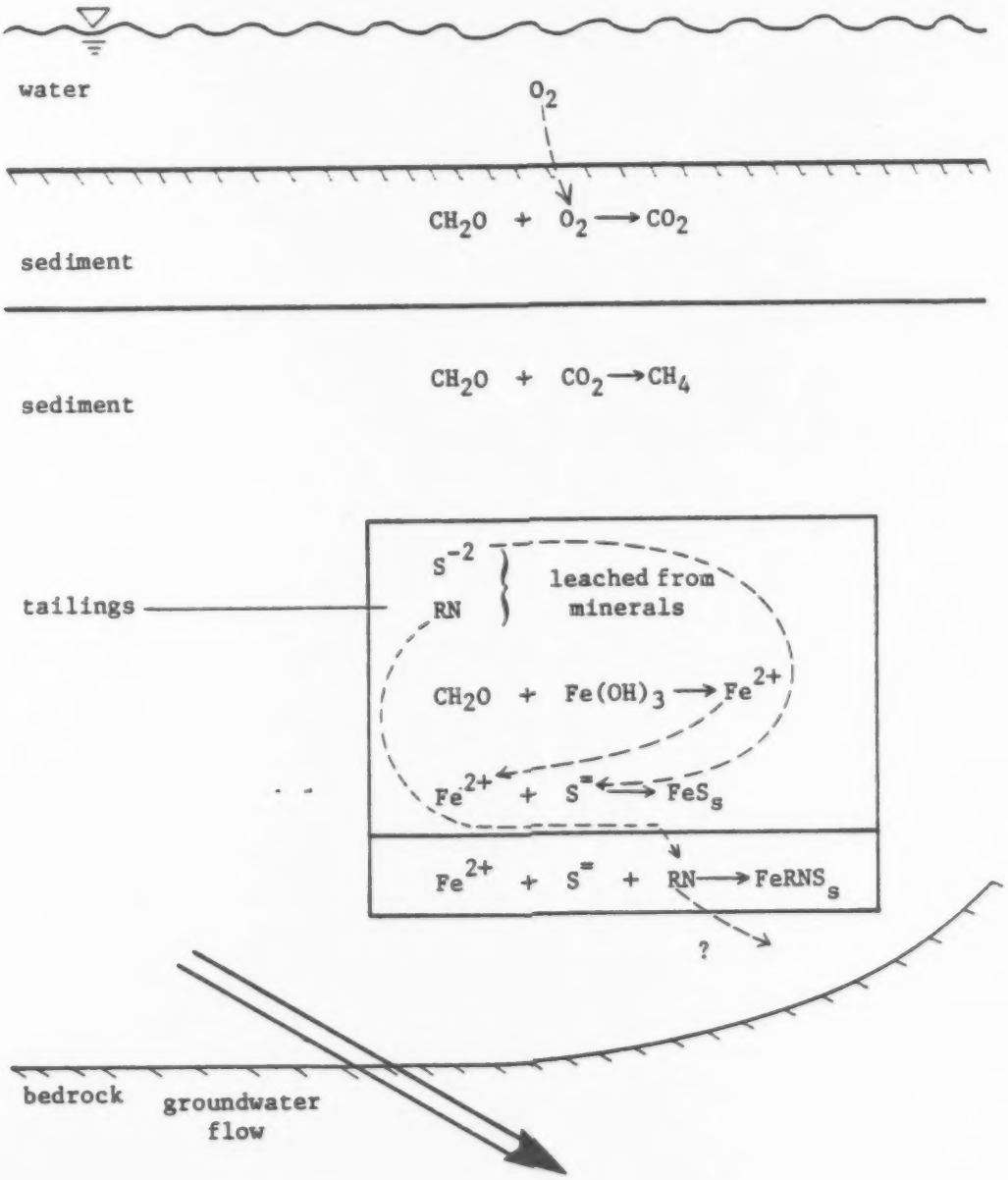


Table 4.1: Summary of major biochemical mechanisms and a few associated geochemical mechanisms as a function of Eh

Relative Eh of tailings system to lake water and sediments for different tailings characteristics			Biochemical system (viewed from point of view of chemical thermodynamics)				Possible Fate of Metals			
Tailings with organics and sulfide leaching	Tailings with sulfide leaching but no organics	Tailings with no organics or sulfide leaching				More strongly oxidizing Eh	Type of Zone	Mobile	Non-Mobile (based on potential for precipitation)	Other potential mechanisms to immobilize soluble metals
Eh	Eh	Eh	Lake Water	CH_2O + O_2		CO_2				
				HS^- + O_2		SO_4^{-2}				
				Fe^{2+} + O_2		$Fe(III)_3$	Aerobic	U, S, U(VI), Cr	Fe oxide Mn oxide	
				NH_4^+ + O_2		NO_3^-				
				Mn^{2+} + O_2		MnO_2				
			Aerobic respiration	O_2 + CH_2O		CO_2	Aerobic, Onic	U, S, V, Cr	Fe oxide to. oxide Mn oxide control	Adsorption Bioconcentration
			Denitrification	NO_3^- + CH_2O		N_2	Anoxic			Bioconcentration
	Sediments Above and Below	Tailings Zone	Mn(IV) reduction	MnO_2 + CH_2O		Mn^{2+}		Fe^{2+} Zn Mn^{2+} Cd Hl Cu	U(IV)	Adsorption Bioconcentration
		Sediments	Fe(III) reduction	$Fe(OH)_3$ + CH_2O		Fe^{2+}	Anaerobic		FeO, MnO, CdS, CuS, HlS, CuS, AsS to. sulfide control	Complexation with organics
Tailings Zone	Tailings Zone		SO reduction	SO_4^{-2} + CH_2O		HS^-		Hl, Cu		Coprecipitation of radionuclides with minerals (e.g., uranite) being formed geochemically
	Sediments Above and Below Tailings		CH fermentation	CO_2 + CH_2O		CH_4	Fermentation Fixation			
			Nitrogen fixation	N_2 + CH_2O		NH_4^+				
					More strongly reducing					

In the lake sediments, bacterial reduction of compounds (O_2 through SO^{-2} to N_2) is theoretically thought to occur as the sediments become increasingly reducing. In each case, one can conceive that one has a layered sediment. In the surface aerobic layers O_2 reduction occurs. In the next layer nitrate reduction occurs followed by MnO_2, $Fe(OH)_3$, SO^{-2}, CO_2 and N_2 in successively deeper layers. In each layer, one type of specialist bacteria exists which is able only to work on one particular type of compound. Whether each layer will develop or not depends upon the relative nutritional requirements of the bacteria and the rate of transport of species to be reduced to the appropriate layers. Such a sediment layering is not found in all sediments. It is a theory; however, sufficient layering is found in lake sediments to allow one to use this as a conceptual base for understanding the possible biochemical controls.

On the left-hand side of Table 4.1, the relative Eh location of three possible tailings masses is shown. The first, tailings with no organics or sulfide leaching would probably border anywhere from anoxic to anaerobic, depending upon the Eh of the groundwater. The second example would be in the same Eh zone as that of the sulfate reduction, except it would be geochemically controlled. In the third example (case III), the redox potential would be about the same as that of methane fermentation.

On the right hand side of Table 4.1, the mobile and non-mobile forms of different heavy metals and radionuclides are given. In aerobic conditions, N, S, U(IV) and Cr are mobile while iron and manganese oxides precipitate; under anaerobic conditions, the reduced metals are mobile except when sulfide is present (causing metal sulfide precipitates). As well, other mechanisms in lake water and in the sediments which can sequester metals and radionuclides include adsorption, bioconcentration and complexation. Of all mechanisms it is expected that precipitation-dissolution reactions and adsorption reactions will be quite important for radionuclides; the possible coprecipitation of a radionuclide with vivanite is raised as a possible reaction. No data were found in the literature describing this possibility.

4.1.1 Perspective on Interactions between Tailings Lake Sediments and Hydrogeology

The direction of leachate transport from tailings is a function of the direction of groundwater flow and of concentration gradients. The actual bleed rate from tailings is a function of the transport rate of material into and out of the tailings, and of the rate of uptake and of

dissolution of materials between the pore water and particulate phases within the tailings mass. Whether geochemical and biochemical mechanisms dominate is a function of transport and of the relative rate of uptake and dissolution by geochemical and biochemial mechanisms. They are influenced by the amount of a material initially present (e.g., organic matter and sulfides for Eh, carbonate or sulfides for pH) and the amounts of these materials being transported in.

Calculations of the relative dominance of different mechanisms can be made if transport rates and mineralogy are known. It is our assessment that one's confidence in predictions of precipitation-dissolution reactions is much less than one's confidence in predictions of transport itself. It is probable that advective transport will be most significant in bedrock and diffusivity most important in sediments in most lakes.

In the processes of mining and milling an ore and subsequent disposal of tailings, the transport rate of an isotope is transformed from one of the order of 10^{-12} cm^2/sec (solid diffusion rate) to one with a spectrum from 10^{-7} cm^2/sec (pore water) through 10^{-10} cm^2/sec (chemical precipitates). This means that radionuclides are more secure in the host matrix of tailings particles than in concrete, if significant dissolution does not occur because the diffusion rate in concrete is similar to that of a chemical precipitate. Accordingly, in addition to precipitation-dissolution reactions, it is expected that a varying set of transport rates will exist in the tailings. Over the short term and maybe over the long term biochemical reactions may control geochemical reactions and hence the amount of material available for transport. Over the long term, if insignificant organic matter is present, geochemical mechanisms may dominate uptake-dissolution reactions. At some point in the long term, solid-state diffusion may dominate over uptake-dissolution reactions; in this case, solid-state diffusion becomes the mechanism which controls transport out of the tailings.

If deep bedrock exists below the lake with a characteristicly low advective flow and a sufficiently high assimilative capacity, long-term flow should be directed to this stratum by sinking several holes through the lake bottom, (e.g., located at a 40 m depth) to such a strata (e.g., located at a 200 m depth) and/or by insertion of a tailings barrier above the tailings. With a low enough permeability, the barrier will direct more flow to deep groundwater than to surface waters or to surface groundwater.

5.0 DRAINAGE BASIN

The drainage basin is the fundamental hydrological unit directing runoff waters into a lake. It is the unit used for various planning functions; it typically is not the unit used for political boundaries.

In the first subsection of this section, the basic characteristics of a watershed and its natural or disturbed terrestrial ecosystem are examined. The concept used is that the basic natural vegetative cover on a watershed is the terrestrial ecosystem. The term terrestrial is used here in its broadest sense. It includes prairie grass on the prairies, lichens in the high arctic, mixed and coniferous forests on the Canadian Shield and mountain areas, etc. It is influenced by resident animal populations and by soil conditions. Modification by disturbances including recreational use, cottages, agricultural and urban areas are considered here and in section 7.1. This delineation attempts to separate out future changes due to man's actions from existing watershed characteristics.

The terrestrial ecosystems modify physico-chemical tendencies of the watershed (vs. if it was not covered). Without any vegetative cover, weathering reactions break bedrock down into soil materials; elements are solubilized and removed from both soils and bedrock via surface and groundwater runoff. With a terrestrial ecosystem present, there is a significant cycle of matter between soluble and particulate phases. The input of matter from the atmosphere is typically much smaller than that of the internal watershed cycle of matter (see section 6.2); however, without this input, this cycle would decrease to a fraction of its existing size.

In this first subsection, the basic character of the watershed is discussed and parameters for measuring these characteristics are examined. This forms a basis for examining relationships between watershed characteristics and lake water characteristics in the second subsection, section 5.2. For a time scale of a year and for fairly large watersheds, the relationships in section 5.2 form the present-day technological basis for examining the driving force for the hydrological and material regimes of lakes. These relationships have just evolved in the past decade; their state of sophistication is still developing.

The technology used as the basis of section 5.2 is inadequate for examining long-term problems or problems of a very short time-scale.

For both short-term and long-term predictions, the effects of the atmosphere upon both the drainage basins and lake must be determined.

Aspects of these relationships are examined in Section 6.2 and 6.4. A short-term example is that a rainstorm (e.g., a 6-hour storm) may cause both large bottom lake currents capable of eroding the tailings in a lake and the erosion of land-based soils which enter the lake. The eroded drainage basin soils may adsorb tailings-released radionuclides and settle out of the lake water, resulting in no net release to the lake water. A long-term example is that acidic precipitation may mobilize more materials out of the watershed than at present, raising the "background" concentrations of these materials (e.g., Fe, S^{2-}).

The concept of the watershed as driving the material cycles of lake processes is an excellent approximation where the atmospheric deposition to a lake surface is 20% or less of that from the watershed, where one is concerned with a relatively short period of time (e.g., a decade) and where the minimum time scale of concern is a season. Where one is concerned with the long-term, the "watershed as the driving force for lake processes" concept is quite a useful building block, but one must not lose sight of the effect that the atmosphere has on the watershed-lake unit.

5.1 TERRESTRIAL CHARACTERISTICS

In this section, the main characteristics of a watershed covered by vegetation are analyzed. For long-term estimation purposes, it would be useful to analyze the basic characteristics of the natural watershed ecosystem before it was disturbed and then to assess its actual present characteristics by including the effects of man. In many regions, such assessments can be accomplished by comparing present watersheds with undisturbed watersheds some distance away. However, such comparisons should be viewed in a qualitative way, and used for planning purposes or scenario testing rather than used as precise measurements because of the difficulty of transposing numbers from one watershed to another similar watershed. As the technology for estimating watershed characteristics evolves, such transpositions may be made in the future with increasing confidence due to narrower error bounds on the numbers and a better understanding of methods for predicting watershed export of material.

The scale of complexity used in assessing the characteristics of the terrestrial ecosystem may be either to describe its internal cycles and, hence, its export (e.g., Likens et al, 1977) or simply to describe its export as a function of watershed parameters ignoring internal watershed cycling between soil and vegetation. Either approach is useful; due to the complexities of internal watershed cycles, it is probable that relationships between watershed characteristics and export will be the most useful tool for the immediate future.

Possible measurements of watershed characteristics useful for describing export of material are the erosional nature of the watershed, surface runoff potential, chemical mobilization factors and terrestrial productivity.

5.1.1 Erosional Nature of Watershed

Possible Measurements

1. various parameters in universal soil loss equation (USLE)
   a) volume of runoff
   b) soil erodibility factor
   c) slope, length and gradient factor
   d) crop management factor
   e) erosion control practice factor

Factors such as those used in the Modified Universal Soil Loss Equation are most useful for assessing the erosional nature of the watershed. Its use is described in more detail in Kilborn (1980); its limitations for application to this work are described in section 6.4. (The limitations are concerned particularly with the fact that it has evolved from agricultural applications).

The volume of runoff describes the erosive potential of runoff. The soil erodibility factor is based on soil properties including sand content, organic matter, permeability and soil structure and is based upon a standardized set of U.S. Soil Conservation Service measurements of soils. The length factor is a function of runoff length and slope. The crop management factor and erosion control factor reflect the effectiveness of different agricultural practices for minimizing erosion.

For application to forested watersheds, all factors used will have relevance; the main need will be substitution of the crop management and erosion control factors with factors and descriptions which either have a broader range (e.g., forest through to agriculture) or which are specific to particular kinds of watershed cover (e.g., agriculture, forest, marsh, etc.). The former approach is useful for a broad based study while the latter is more useful for a particular kind of watershed. Further, the soil erodibility factor would need a broadening from being strictly porous media to include the spectrum of soils to bedrock.

Similar equations are evolving in the literature for other kinds of ecosystems, but none appear to be as well developed as the USLE. It is suggested, then, that the factors of the USLE be viewed in the conceptual sense - they have an excellent utility for describing erosion potential; their actual application will need some care.

5.1.2 Surface Runoff Potential

Possible Measurements

1. various parameters in USLE
2. precipitation-runoff relationships
3. runoff coefficient

The surface runoff potential describes the potential quantity and pattern of runoff. It is highly dependent upon antecedent conditions (e.g., soil moisture before the start of rainfall) for any one given event. Over the long term, antecedent conditions even out, in which case the runoff potential becomes a function of more macro properties such as changes in type of soil cover and long-term changes in rainfall volumes.

As a rainfall commences, whether or not it will be transformed into runoff depends upon the ability of ground to absorb water. Such an ability is measured in part by factors in the USLE. This ability also depends upon the degree of saturation, and whether or not the ground is frozen (antecedent conditions). When the infiltration potential is saturated, the extra rainfall (termed excess rainfall) is transformed into surface runoff.

As an indication of runoff potential, use of rainfall statistics are appropriate. These statistics are described further in section 6.1 and are reviewed here. Precipitation records provide the data base from which the frequency and magnitude of return rainstorm events may be predicted. The Log Pearson Type III method of distribution is used to predict various frequencies of return storms such as 2, 5, 10, 25, 50 and 100-year. Besides using semi-empirical methods, they may also be based on actual storm experiences (Kilborn, 1980). Some rainfall events used in runoff forecasts include:

Area	Event
Northern Ontario	- Timmins Storm
Southern Ontario	- Hurricane Hazel
Eastern Ontario	- 100-year storm, or Timmins Storm, whichever is greater
Saskatchewan	- 100-year storm or July 24, 1976 (a 12-inch storm)

In large mountainous watersheds, the actual stream flow probability is more pertinent than the rainfall event. British Columbia, for example, has designated the 200-year flood flow as a design parameter for tailings sites near major rivers.

Various mathematical models are available for calculating the detailed structure of runoff as a function of rainfall and of surface characteristics. They generally give excellant estimates of actual runoff as opposed to runoff potential. Their use is considered further in Kilborn (1980) and in section 6.2. From such models, one can extract information relevant to describing runoff potential or excess rainfall. The following discussion is extracted in part from Kilborn (1980). Its relevance is mainly associated with estimation of peak runoff potential rather than average annual runoff patterns.

Excess rainfall is the portion of rainfall resulting in runoff. The value is determined by subtracting the portion of rainfall infiltrating the ground from the total rainfall. The infiltration amount may be estimated using SCS techniques based upon the Hydrologic Soils Group, runoff parameters (representative of the surface cover) and other parameters. Soils are classified into Hydrologic Soil Groups as follows:

A Low runoff and high infiltration when thoroughly wetted
B Moderate infiltration when thoroughly wetted
C Slow infiltration when thoroughly wetted
D Very slow infiltration and high runoff when thoroughly wetted

Snowmelt also is a measure of runoff potential. Watersheds in northern climates are subject to spring flooding from snowmelt from the approximately six-month average snow pack accumulation over the winter. Large spring runoffs are typically caused by warm spring rains absorbing snow (snow melt) before the ground loses all of its frost. The snow cover in the basin must be known to predict runoff volumes and peak flow rates. Methods currently used for predicting snowmelt are listed below in order of increasing complexity and refinement:

i) Temperature Index or Degree Day Method,
ii) Degree Days plus Recession Analysis (U.S. Bureau of Reclamation),
iii) Generalized Snow Melt Equations (U.S. Corps of Engineers),
iv) Index Plots plus Regression Analysis,
v) Hydrograph Synthesis plus Streamflow Routing.

As a coarse indicator of runoff potential, an assumption that a large fraction of the snowpack on sloping land (e.g., 80%) becomes runoff is reasonable, provided the ground remains frozen and insignificant quantities are trapped by marshes.

A final measure of runoff potential is the areal runoff coefficient. It is an actual quantification of the average runoff volume per unit area over a decade or longer. It is obtained by using data from gauging stations, obtaining the unit runoff coefficient for each station and interpolating between gauging stations to give lines of equal runoff. It is typically summarized in the form of maps containing iso-runoff lines. They are based upon present conditions; rainfall-runoff relationships are required to make inferences about future changes.

5.1.3 Chemistry of Surface Runoff

Possible Measurements

1. major ions in surface water runoff
2. leachate from major bedrock outcrops
3. leachate from major soils

Chemical mobilization factors are those mechanisms which control the chemistry of runoff for different watersheds given possible shifts in the future chemistry of rainfall.

Major ions in surface water runoff give a measure of the present mechanisms dominating mobilization; inspection of major landforms (bedrock outcrops, soils) and of the present terrestrial ecosystem, and use of correlation analyses between rainfall chemistry and these landforms permit statements regarding present mechanisms.

For predicting future conditions, inferences are needed concerning how present mechanisms could potentially change. It is suggested that the leachability of present major soils and of presently-exposed or potentially-exposable bedrock and the types and quantities of leachates are possible measures of chemical mobilization factors. Such information can be inferred qualitatively from knowledge of rainfall chemistry and the chemical character of bedrock and watershed soils.

5.1.4 Terrestrial Productivity

Possible Measurements

1. gross biomass production
2. biomass producton of particular plants
3. area of ecosystem based upon dominant plants communities
4. influence on weathering reactions
5. net organic export to lake system

Terrestrial productivity provides an indication of the viability of forest cover, the extent of watershed vegetative cover and the catalytic or mitigating effects upon chemical mobilization due to the terrestrial cover. It can have a catalytic effect by mobilizing certain elements from soils which would normally be released only at slow rates through the processes of uptake from the soil, and by formation of biomass and subsequent decay. It can have a mitigating effect by protecting the soil from water-driven erosion.

Possible measurements such as gross biomass production (of all plants) and biomass production of particular plants give indications of the viability of the vegetation. The area of watershed covered by dominant plants (e.g., marsh ecosystem, prairie ecosystem) provides a more macro

level of information than "production" estimates. It provides a useful measure for incorporation into the "erosional nature of the watershed" and for assessing "chemical mobilization potential".

5.1.5 Perspective on Terrestrial Characteristics

Terrestrial characteristics describe the potential materials which may be released from a watershed to a lake and the effects that the terrestrial ecosystem has upon this release. The release of particulate materials (erosional nature of watershed) has received wide study, particularly for agricultural areas; use of broadened concepts are needed and are evolving in order to examine all possible terrestrial ecosystems. Surface runoff potential has received extensive study; adequate knowledge appears to be available for consideration of a deep-lake disposal application. The chemical mobilization characteristics and the associated effects of a terrestrial ecosystem have received relatively less study. Inferences can be obtained from a knowledge of gross watershed character (mineralogy, terrestrial cover) and the chemistry of surface runoff. Methods such as relationships of watershed export resulting from atmospheric deposition, which are examined in section 6.2, allow more quantitative inferences concerning chemical mobilization.

For the long-term, use of macro-parameters (average annual characteristics), as opposed to micro-parameters, is probably most appropriate for examining long-term, average characteristics of a lake. Micro parameters strongly influence the actual extent of extreme runoff or extreme erosion; however, one would need to question their actual utility for examining questions relating to potential lake disposal of U-mine tailings. Where available knowledge is sufficient, their selection may be appropriate.

When a decision concerning the "micro to macro" character of the parameter is made, then appropriate tools should be used for examining each parameter. For example, if it is decided that average annual runoff is the appropriate parameter for describing surface runoff potential, then use should be made of the areal runoff coefficient, rather than of a detailed rainfall-runoff model which describes the runoff pattern of one storm. One caution is in order - if the long-term rainfall picture will change, then a macro rainfall-runoff model would be needed to estimate the change in the areal runoff coefficient.

5.2 DRAINAGE BASIN-LAKE INTERACTIONS

In this section, relationships between drainage basin export and lake characteristics are examined. For certain materials in the lake, drainage basin export is the main driving force. This depends partially on the relative amount of atmospheric and drainage basin input. Materials for which drainage basin influences dominate are considered here. Some effects of the drainage basin on other materials are considered here, but they are treated mainly in section 6.4.

5.2.1 Hydrological Budget Aspects

Possible Measurements

1. average annual lake inflow
2. high-flow extrema
3. low-flow extrema

In many lakes which have a large drainage basin and small surface area, surface water inflows are the major portion of the hydrological budget. In other lakes to which rainfall approximately equals evaporation and to which material inputs from the atmosphere are small relative to inputs from the watershed, the hydrological budget can usefully be described by considering only the surface water inflow. In all other lakes, surface water runoff is only one pathway (albeit a critical pathway) for entry of water into the lake.

Runoff characteristics into a lake are generally described by three statistics: mean annual runoff, high flow extrema and drought extrema. The mean annual runoff may be expressed as an annual quantity (e.g., $km^3/yr.$, m^3/sec) or as a unit runoff coefficient ($km^3/yr.$ per km^2, a height of water per unit area of the watershed). Maps of unit runoff characteristics in the form of isopleths are available for most areas of Canada. They permit estimation of long-term runoff characteristics of fairly large-sized watersheds. High flow extrema describe the frequency (e.g., once in 100 years) with which a given size of runoff is expected to be equalled or exceeded on a long-term basis or the runoff's Return Period (the time interval between runoff events of equal size, e.g., a 100 year storm). Low flow extrema describe the frequency with which a certain magnitude of low flow will be equalled or exceeded (e.g., a drought flow, described as a 7-day 10-year low

flow, is the 7-day average low flow which will be exceeded once every 10 years). These extrema have been shown to fit a particular probability distribution (partly on a theoretical basis, partly on an empirical basis). The log-normal distribution is used for many flood events.

The average annual runoff is the common time base for examining variations in long-term characteristics and for calculating the lake detention time (the time it would take to fill the lake at present rates of inflow). The latter is a measure of a short-term extreme; it is more useful as an indicator of watershed behaviour than of lake behaviour, provided that the volume of water associated with the flood flow is small compared to the volume of the lake.

5.2.2 Material Budget Aspects

Possible Measurements

1. average annual input to lake (e.g., kg/yr.)
2. areal lake loading (e.g., gm/m^2-yr.)
3. lake loading-concentration models

For many lakes, inputs from the drainage basin are the main source of supply to the lake. For lakes whose watersheds have been significantly influenced by man, this is particularly true. For other lakes, watershed inputs are typically as important as atmospheric inputs.

For those lakes to which groundwater and atmospheric inputs are small, and whose watersheds are relatively undisturbed by cultural effects, watershed erosion and mobilization of materials are the source of "background".

The average annual input to a lake is one measure of flux to a lake, while the areal lake loading (equal to the average annual input divided by the lake surface area) is a parameter dimensionalized to the size of the lake. The latter measure attempts to show the effect of the same size of input upon two different sizes of lakes.

For lakes to which atmospheric loadings are small and whose watersheds are relatively undisturbed by cultural influences, the average annual input will approximately be equal to the background concentration times the average inflow volume of water. This provides a measure of the

rate of mobilization of materials such as radionuclides from the watershed. In particular, for areas in northern Canada where background levels may be dominated by leaching of exposed veins, it represents a measure of the rate of leaching, if it can be attributed solely to certain watershed areas. Although average annual mass input to lakes provides a practical indication of mass fluxes, it must be recognized that natural lake basin cycles occur on several levels and all of these levels must be considered in an evaluation. For example, the first level may be diurnal, ng for example night-day thermal variations, the seasonal reflecting longer term thermal variations, may relate to forest fire burn frequency reflecting lar ale erosional and nutrient release cycles. Beyond this there may be longer term climatic cycles.

The mass flow of material into a lake is a critical parameter for assessing permissible rates of leaching out of tailings into a lake - a fraction of it may be called an acceptable bleed rate.

Some caution must be used in applying estimates of lake inputs. Part of the export occurs as a dissolved phase, part occurs as an eroded particulate phase. If significant export from a watershed occurs via bedload transport (e.g., shifting sand/clay grains along the stream bottom), it will typically not be measured because such material needs a special sampling device. If materials such as aluminum are located in the matrix of minerals, they may not be measured unless chemical extraction procedures are able to break up the matrix. Both errors act to cause underestimates of export.

A lake loading-concentration model is the relationship which relates lake concentration to the input rates. The relationship may be empirical or based upon the law of conservation of mass. Empirical relationships involve relating lake concentrations to such factors as lake loading (the average input rate to the lake), lake mean depth, and lake detention time (e.g., Kirchner and Dillon, 1975; Reckhow, 1978 for phosphorus). Law of Conservation of Mass-based relationships relate lake concentration to the loading, outflow rate, lake volume and any internal loss and recycling mechanisms. They vary from fairly simple (e.g., Dillon & Rigler, 1975) through more complex (e.g., Snodgrass and O'Melia, 1975) to quite complex (e.g., Thomann et al, 1975). The utility of various degrees of complexity is assessed in articles such as Orlob (1975) and Snodgrass (1978).

While these kinds of models have been most widely applied to the nutrient analysis of lakes, their form and their capabilities and limitations are equally applicable for the estimation of all other materials, including radionuclides. The only differences are the mathematical expressions required to describe internal lake cycling mechanisms between particulate and soluble phases. Such mechanisms are discussed previously in section 2.1.

In the use of such models, adequate examination of the data base, lake characteristics and probable internal lake mechanisms (between soluble and particulate phases) is needed before a model is selected. These points are further discussed by others (e.g., Orlob, 1975; Snodgrass, 1978).

These models provide a basis for determining the long-term chemistry of lake water. Of particular use would be relationships which relate the potential stripping ability of a lake for removing tailings leachates which may enter the water column from the lake sediments. Potential relationships between land-based inputs of organic carbon and their stripping ability are discussed here; relationships between inputs of nutrients and algal regimes and their stripping ability are discussed in Section 5.2.3.

Within most natural fresh waters, the range of organic carbon commonly falls in the range of from 0.1 to 10 mg/L carbon (Stumm and Morgan, 1970). It is derived principally from two major sources: 1) allochthonous sources through both the direct input of organic materials such as in the form of autumn shed leaves (Kaushik and Hynes, 1971), and indirect inputs of soil organic material from sheet and shallow groundwater inputs; and 2) from autochthonous sources such as the cellular debris and extracellular excretion products of aquatic organisms (Fogg, 1966; Leppard, 1977).

The chemistry of this organic material is varied and complex. The nature of simple soluble exudates such as glycolic compounds from algae has been described by Fogg (1966), while others such as Leppard (1977) have described the morphology and, to a limited degree, the chemistry of algal and bacterially-derived fine particulate organic matter, which appears to consist of up to 20-30% urionic acid residues.

From the terrestrial system, much of the direct particulate inputs of organic matter to freshwater systems provide for nutritional require-

ments of bacteria; these components are prinicipally carbohydrates and protein (Kaushik and Hynes, 1971). Within the soil systems that supply organic leachates to lake and stream systems, humic substances constitute the bulk of the organic matter. They have been described (Schnitzer, 1971) as acidic, dark coloured, partially aromatic, and chemically complex substances with molecular weights ranging from a few hundred to several thousand. They lack specific chemical and physical characteristics (e.g., melting point, refractive index, exact elementary composition, etc.) usually associated with well-defined organic compounds. Based upon their solubility, Schnitzer (1971) has suggested that humic materials may be further broken down into: 1) fulvic acid, which is soluble in both alkali and acid and is considered to have the lowest molecular weight; 2) humic acid, which is soluble in alkali but insoluble in acid and of intermediate molecular weight; and 3) humins, which are insoluble in both alkali and acid and have the highest molecular weight.

Dissolved organic materials first begin to influence the biotic nature of the watershed in the headwater areas where rains and melting snow provide the water necessary to transport the organics from this place or origin. In the soils and wetlands of the watershed, vegetation releases a wide variety of soluble and insoluble, but relatively simple organic compounds. Many of these are used by a wide variety of soil and sediment fungi and bacteria. These soluble organic materials are, to a large degree, the products of soil and aquatic microbial and fungal populations. As they move out of the shallow soil horizons and organic rich aquatic sediments, and as they are carried through shallow groundwater and surface stream systems, they come in contact with the minerals of the parent glacial till or bedrock.

As early as 1929 Harrar recognized the ability of organic acids to leach, reduce and dissolve minerals. Since that time the role of naturally-occurring organic acids in the dispersion of many elements has been widely recognized. (An excellent review of the role of natural organic acids in the dispersion of a wide variety of elements has been published by Jackson et al, 1978.) Organic acids such as fulvics react with many of these metals to form stable complexes. These stable complexes are transported throughout the watershed carrying a wide variety of trace elements and nutrients. The nature of the bioavailability of many of the elements, complexed or otherwise bound to the soluble organic materials in water, is not well known.

For management purposes, it would be useful to have relationships available which relate loadings of organic carbon to lake concentration of such trace elements as radionuclides. It is clear that, due to the sequestering ability of organics (particularly organic acids) for trace metals, such relationships exist in nature. Unfortunately, due to the lack of sufficient knowledge, such relationships have not been quantified. Further, as one attempts to include more processes in quantitative relationships and to describe shorter time-scales, knowledge of these processes becomes one limitation to the formulation and use of such relationships. Literature synthesis (of data from many watersheds) may provides a partial basis for such relationships. Such relationships are the tool required to assess future lake behaviour for those lakes and heavy metals in which organics dominate the sequestering and removal of trace metals from the lake water.

5.2.3 Material Budget Relationships to Biological Regimes

Possible Measurement

1. lake loading-biomass models

As indicated in section 2.1, the biota of a lake is functionally dependent upon its water chemistry and vice versa. Many relationships exist between lake biota and water chemistry (e.g., relationships of chlorophyll a to total phosphorus such as those of Dillon and Rigler, 1975). Relationships exist which relate water chemistry to lake loading. For several materials, efforts have been made to short-circuit use of water chemistry as a variable and instead of relate lake biomass directly to the actual loading. Such a process eliminates one of the variables involved, but in some cases reduces the generality of the relationship because some lake property (e.g., anoxic hypolimnetic waters) changes the fundamental relationships involved.

Typical models describing such relationships are those of Vollenweider (1968) which relate lake trophic status to phosphorus and nitrogen loading, and those of Yeasted and Morel (1977), of Schnoor and O'Connor (1980) and of Thomson et al (1975) which relate phytoplankton biomass to phosphorus and nitrogen loading. In these investigations, phytoplankton biomass is used as a more optimum measure of the state of eutrophication than total phosphorus. These relationships vary from being relatively simple to fairly complex; the simple relationships are

probably fairly adequate for lake disposal questions but have some limitations.

For algal growth, levels of inorganic nitrogen (NH_3, NO_3), phosphorus, total inorganic carbon, essential trace metals and vitamins influence the rate and extent of growth. For the majority of lakes in Canada, phosphorus appears to be the element controlling the extent of algal biomass, while both nitrogen and phosphorus can control the rate of algal growth at different times of the year (this statement assumes standardizing for light, temperature and turbulent controls on growth). Due to the low levels of inputs of P to lakes and the relative ease of controlling its concentrations, the environmental engineering field has generally opted for phosphorus removal in wastewater treatment plants, arguing that even if nitrogen is presently limiting the extent of algal biomass, phosphorus can again be made to be limiting by lowering its input rate to the lake water. In seawater and estuaries, nitrogen appears to control the extent of biomass development. Hence, for Canadian lakes, use of phosphorus loadings as the driving force for average annual biomass is usually appropriate for present conditions and for foreseeable conditions in the future; however, some care must be used if the relative export of N and P from the watershed changes over time or if significant fluxes of N enter the lake from the atmosphere in the future. Use of the N/P ratio is a useful monitoring parameter here.

For estimates of trace metal stripping from the water column, one may use material budgets, provided that the mechanism controlling the stripping is acknowledged (i.e., whether it is associated with algal remains, organics (e.g., humics and fulvics) from the watershed or with inorganic particulates).

In the simple-mass balance models, the effective settling coefficient of particulate forms of phosphorus is found to be of the order of 10-50 m/yr. (e.g., the models of Dillon and Rigler, 1975; Snodgrass and O'Melia, 1975). The observed settling rates for Th228 and Ra226 are of the order of 10 to 100 m/yr. in lakes in which whole lake addition of these isotopes was made (e.g., Hesslein 1978, 1979; Schindler, 1979). Given the fact that particulate forms of phosphorus are a means of estimating algal biomass on such a lake, and given the similarity in rates, some promise may exist for combining algal biomass - phosphorus loading models with radioisotope models to compute (or to set limits upon) the expected removal rate of the isotope through the process of

incorporation of radioisotopes and subsequent sedimentation. If this hypothesis is proven, then one has obtained a tool for relating radioisotope concentrations to nutrient loading (e.g., phosphorus).

The justification for combining biomass-trace element relationships has been described in section 2.1.5 previously. However, a caution is in order - another parallel relationship may exist. Such a relationship might be caused by the association of heavy metals with organic molecules such as humic and fulvic acids; parallel levels of inputs of phosphorus and organics to the lake from the drainage basin and the association of a heavy metals with land-based organics might produce the parallel relationship. However, given the low settling rates of many organic molecules, such a parallel relationship is probably not important unless trace-metal associated organics attach to cellular detritus or algae.

5.2.4 Transport Upset due to Extrema

Possible Measurements

1. fingerprint features in lake sediments
2. seismic activity

The long-term transport of material from a drainage basin to the lake can be changed by extreme events such as rock slides, earthquakes, and meteorologically driven erosion. Such upsets could increase the rate of accumulation of sediments or block up normal upstream inputs of material by altering the channel of influent rivers.

Past upsets can be occasionally traced by examination of lake sediments, using them as a fingerprint to the past. The probability of future upsets can be inferred by correlating the likelihood of an upset with the stability of terrain of the watershed. Quantification of the alteration can probably only be made in a realistic manner by scenario testing (e.g., What if a certain river is blocked off in the future? What is the relative effect if the tailings were upset by the same cause?)

5.2.5 Perspective on Drainage-Basin Interactions

Unifying ideas for relating drainage basin inputs to lake concentrations are evolving. One used by Snodgrass (1980) and advanced by Turekian (1977) is used here as a method for presenting one such attempt to develop a set of unifying ideas. While the emphasis in this attempt is upon the estuary-oceanic areas, it is useful for lake purposes. For the purpose of obtaining management-level parameters, it further points a direction for the summarizing of data from databases and studies such as those of PLUARG.

In a presidential address before the Geochemical Society, Turekian (1977) noted that calculated theoretical concentrations of nickel and other trace metals in oceans are orders of magnitude higher than measured values. These calculations are based on in vitro experiments and homogeneous reaction kinetics. He believes that the answer lies in the role of particles, acting as sequestering agents for reactive elements as the particles are transported from the continents to the ocean floor. He proceeds to describe a five-step conceptual model which describes the controls on distributions of many trace metals as particles are transported from the continents to the oceans. He uses evidence from radionuclides and diagnostic stable nuclides to support his model. Radionuclide evidence is useful support, provided the ratio of radionuclide concentration to the concentration of the trace metal of concern remains constant.

Turekian's model may be taken as a framework for summarizing the various controls of trace metal fragments. Many of Turekian's ideas have not been proven irrefutably, but they are on the whole supported by literature. Turekian's model is as follows:

1. Soils and weathering profiles retain many trace metals supplied to them by the atmosphere or by weathering.

2. Metals are further sequestered by particles during stream transport.

3. Further removal of particles and their associated elements takes place at the stream-ocean boundary where a few elements are released.

4. The estuarine trace metal cycle is mainly self-contained: release and deposition occur virtually entirely within the system.

5. The major sequestering of trace metals in the oceans is mediated through particles.

The premise that soil and weathering profiles retain their source characteristics is supported for many major elements. Use is made of ratios of elements in suspended particulates as one method of examining this question. For streams which have ratios higher than continental soils, one may attribute such to element-rich soils or to industrial contamination. Such enrichment is a fingerprint tool - are there unknown lobes of a material in the watershed being leached by weathering reactions?

In lakes, fewer detailed comparisons have been made between lake suspended matter or lake sediments and weathering characteristics; rather, attention has often been directed to attributing sediment concentrations to industrial and urban influences. Since lakes act as settling basins for particulates, it is expected that their sediments should reflect the characteristics of weathering products, despite redistribution by bottom currents.

The premise that "metals are sequestered by particles during stream transport" has been shown for certain metals; it requires more work for others. The major problem in streams is to determine the sequestering mechanism. As Jenne (1968), Hem (1977), and Turekian (1977) note, the hydrous metals of iron and manganese have a large capacity for sorption and coprecipitation with trace metals. Such hydrous metals exist as coatings on inorganic particles, particularly clays, and hence have a much greater capacity for capturing trace metals than their weight per unit volume would suggest. Correlations between metals and iron have been found by several investigators in lakes and in suspended phases in rivers, but direct correlations have also been found with organic matter. Turekian (1977) noted that in the presence of only hydrous oxides, one might expect that the system would be purified of trace metals by reaction with the oxide. Organic matter such as humics can compete with hydrous oxide surfaces for trace metals to form a group such as a nickel-organic complex. The competition would be expected to be particularly great in sediments, unless the nickel is essentially irreversibly adsorbed or coprecipitated on a hydrous oxide surface. The competition between hydrous oxide surfaces and organics for metals is at present a major area of experimental and modelling research.

Some environmental evidence supports the notion that, generally, hydrous oxides of iron dominate organic matter in competing for trace elements such as nickel. It includes associations between metals and iron in coatings scraped from lithic fragments and quartz pebbles in rivers. Such coatings are probably found where trace metal and iron-rich anoxic groundwater interfaces with oxic stream water. The best evidence for dominance of coatings in the natural environment is Gibb's (1973, 1977) data which suggest that coatings outcompete organics globally. Further, export of many metals to the oceans as a soluble form is less than 20%. However, whether coatings dominate in small watersheds which have large releases of organics and negligible export of inorganic particulates is unknown.

For managment purposes, then, the qualititative model of Turekian (1977) provides a conceptual base for examining long-term drainage basin-lake interactions. Particulate export dominates transport for many metals and, hence, presents sites for further sequestering in the lake. Whether the transport and, hence, the potential uptake mechanism in the lake is due to association with coatings or with organics is a matter of debate. For lake disposal questions, it is suggested that use of an empirical parameter such as the dominant particle exported (i.e., whether the lake watershed is dominated by export of inorganic particulates or of organics) may be a reasonable tool for viewing these complex interactions. Then, for internal lake uptake of potential tailings leachates by particulates, one may examine the relative magnitude of inputs of organics (e.g., humics), inorganic particulates (e.g., hydrous oxide coated clays) and phosphorus (as a measure of the development of biomass) and their associated ability to sequester the potential tailings leachate. This is a suggestion which is useful, and conceptually, for asking what is most important. Its application to actual practice requires further work.

One further caution is in order. For some elements (e.g., CO_3, Cl^-) transport from the drainage basin to the lake occurs primarily in a soluble form. The ratio of soluble to particulate transport to a lake from its drainage area provides a parameter for determining the relative importance of each mode of transport.

6.0 ATMOSPHERE

Atmospheric characteristics and long-term behaviour influence the long-term characteristics of the watershed and of the lake. In this section, existing characteristics and possible long-term behaviour are sketched. Then, the interactions between the atmosphere and watershed and between the atmosphere and lake are examined. It is through the former that the actual mobilization of contaminants and erosion of soils and bedrock occurs. It is through the latter that long-term mixing regimes of lakes are determined. Then the long-term behaviour of the atmosphere as a driving force for interactions between the drainage basin and the lake is examined. It is effects of the atmosphere as a driving force for watershed-lake interactions that produce the long-term background concentrations in lake water. Assessment of background concentrations together with assessment of the incremental concentrations due to transport of tailings leachate to the lake water are the tools necessary to assess the propriety of a deep-lake disposal option for U-mill tailings.

6.1 ATMOSPHERIC CHARACTERISTICS

Five atmospheric characteristics which have important influences upon a lake, its drainage basin and its hydrological cycle are radiation, wind, rainfall, temperature and atmospheric chemistry (particularly precipitation chemistry). These characteristics are now examined as descriptors of atmospheric properties. Their individual and combined effects on the watershed and lake are examined in subsequent sections.

6.1.1 Radiation Characteristics

Possible Measurements

1. solar radiation incident upon earth
2. solar radiation absorbed by atmosphere
3. atmospheric radiation
4. atmospheric emissivity

It is useful to divide radiation received at a lake's surface into two types of radiation: radiation transmitted through the atmosphere from the sun and radiation generated by the atmosphere itself acting as a

radiating body. The former is called solar radiation, the latter is called atmospheric radiation.

Absorbed solar radiation (radiation absorbed by the atmosphere) is the major heat input into Canada. It is the climatic element which most influences plant growth, hydrologic cycles and daily temperatures. Assuming that a constant amount or intensity of solar radiation strikes the outer layers of the earth's atmosphere, changes can occur during its transmission through the atmosphere due to dust, CO_2, H_2O, O_3 and the presence or absence of clouds. Increased activity on the surface of the sun may vary energy emissions to the Earth, but over a timescale of 5 000 years can be considered insignificant compared to the effects of the variation of the transmissivity of the Earth's atmosphere. Dust particles (of volcanic and agricultural origin) will prevent part of the incident solar radiation from passing through to the Earth's surface, thus promoting cooling of the Earth's atmospheres. Carbon dioxide and water vapour absorb outgoing terrestrial radiation and produce a greenhouse effect. This gives a tendancy towards increasing the surface air temperatures. The oceans, however, tend to act as reservoirs for both heat and carbon dioxide. How the effects of atmospheric dust, atmospheric CO_2 and oceans as a heat sink will interact to influence the atmospheric temperature is a matter of some controversy. However, their combined effects upon radiation characteristics of the atmosphere are not anticipated to be large over the next 10 000 years unless a significant shift in particle content of the atmosphere occurs. This would then have an effect on the total ecosystem; the relative effect upon a deep-lake disposal option is discussed in Section 6.4. Short-term natural shifts in properties such as atmospheric emissivity may be anticipated due to periodic volcanic eruption.

Solar radiation (short-wave) and atmospheric radiation (long-wave) may be measured using standard measurement devices (e.g., an Eppley pyrhelimometer for solar radiation); they may be calculated using suitable formulae (e.g., that of Wong, 1977 for solar radiation; that of Octavio et al, 1977 for atmospheric radiation). These formulae are quite precise - in fact, a formula is often the method of choice for estimating atmospheric radiation. For anticipated changes in such atmospheric properties as clarity due to dust, the anticipated changes in radiation can be calculated with quite good precision. These changes then provide a method for assessing the effects of long- term changes in atmospheric properties upon radiation characteristics.

Emissivity may be measured as an average emittance of radiation from a body. The atmospheric emittance is the ratio of total atmospheric radiation to total black body radiation at the same temperature. It may be calculated by suitable formula.

6.1.2 Wind Characteristics

Possible Measurements

1. frequency diagram of wind speed
2. frequency diagram of wind direction
3. frequency of occurrence of extrema
4. change in global circulation patterns

Winds represent the flow rate of air from one location to another under the influence of pressure gradients. Their patterns and velocities are highly influenced by atmospheric stability, and by the type and movement of air masses. Future patterns are dependent upon future climate.

Climatic change may be induced by a change in the general circulation of the atmosphere. Most of Canada lies in the zone of the Westerlies, but wind directions are dominated by local high and low pressure systems of the westerly stream. Generally, the west coast experiences winds which tend to the west to southwest section; the interior of Canada experiences a west to northwest flow; and the Atlantic coast gets a west to southwest flow. Changes in the general circulation are linked to solar and atmospheric changes, which may or may not take longer than a 10 000 year time frame to be significantly altered.

Wind velocities and directions are recorded on an hourly basis by the Atmospheric Environmental Service of Environment Canada. Frequency diagrams of wind speed and wind direction are methods for summarizing this information. The frequency of occurrence of extrema and long-term change in global circulation patterns are two variables which are of interest here. Meteorological models are tools for predicting these variables; however, their ability to make long-term prediction is problematic because there appears to be a lack of knowledge of all variables which control circulation patterns. Only, for example, when such models are capable of predicting the onset of an ice age is it probable that they will be able to make predictions of long-term changes in global circulation patterns. Hence, estimates of

atmospheric wind characteristics and their effects upon a drainage basin-lake system can be made only when it is clearly understood that uncertainties will remain in these estimates. Clear enunciation of these uncertainties (in qualitative or quantitiative terms) is proposed by BEAK to be a reasonable method for making inferences of long-term wind characteristics.

6.1.3 Precipitation Characteristics

Possible Measurements

1. rainfall
2. fingerprint features as indicators of past rainfall
3. possible future changes in rainfall quantities

Precipitation behaviour is dominated by the solar radiation regime and by local weather systems. Canadians are accustomed to spectacular day-to-day changes which come under the classification of natural variations. However, to significantly alter this "moisture" regime would require a serious upset of the balance existing today. The moisture has to originate from the Earth's surface, and will eventually return. A change in atmospheric circulation may create regional variations, but the probability of occurance of extreme disturbances on a worldwide (and even Canada-wide) basis over the next 10 000 years may not be significant.

At present, rainfall is measured routinely at gauging stations by the AES. Use of the tools of hydrologists and climatologists permit one to estimate rainfall between such stations. The long-term variations in future rainfall patterns may be estimated using meteorological models; however, the problems outlined above (Section 6.1.2) apply here. One reasonable approach is to use estimates of variations over the past several thousand years as a guide to the future. To make such inferences, use of fingerprint features in existing lakes is suggested. These features come from several fields. An example is evidence for variations in lake levels during a 1 000-2 000 year time frame deduced from minerals in a lake's sediments. A second example is the anthropological debate on the ecology of early man on the shores of Lake Geneva (see below, discussion of lake levels).

6.1.4 Thermal Characteristics

Possible Measurements

1. temperature
2. future changes in heat fluxes

The temperature of different air masses influences the movement of air masses and general atmospheric circulation patterns. It is the result of the major heat fluxes between the sun, space and the earth's surface. Small future changes (equivalent to $\pm$ 0.5°C over the globe) may fundamentally change the glaciation pattern, wind circulation patterns, etc.

Estimation of future atmospheric changes contains many unknowns. In particular, it has been hypothesized (Lovelock, 1980) that self-compensating global mechanisms may minimize the effects of regional climate changes such as glaciation upon the whole earth by inducing changes in tropical areas and modifying oceanic and atmospheric circulation parameters. Critical analysis of global self-compensating mechanisms provides one method for analyzing the future accuracy of estimates of atmospheric characteristics.

6.1.5 Atmospheric Chemistry

Possible Measurements

1. relative composition of major gases (e.g., O_2, N_2, etc.)
2. pH of rainfall
3. major metals, anions and nutrients
4. nature of particulates

The gaseous composition of the atmosphere has been measured widely. A reasonable assumption is that the atmospheric composition of many gases such as O_2 is mainly controlled by biological processes (e.g., global production by vegetation and oceanic algae). If such a composition changes significantly in the future, many other problems will arise. Comparison of the actual importance of these problems to those caused by potential leaching of materials from tailings is one method for assessing whether variations in the elemental composition of the atmosphere is a relevant question for tailings management. Policy statements by regulatory authorities as to whether changes in the gaseous composition should be considered by a proponent of a deep-lake disposal option are probably in order.

The pH of rainfall is controlled by several factors. If a water droplet is at equilibrium with the atmosphere, its natural pH is 5.7 and is controlled by dissolved CO_2. Addition of low concentrations of sulfuric acid (fossil fuel burning, ore roasting) and of nitric acid (fossil fuel burning, soil degassing transportation) decreases the pH to the range of 3-5 depending upon the geographical area. The addition of inorganic particles such as clays, aluminum oxides, and limestone provide sites the removal of H^+ from the water droplet and hence, for the raising of the pH.

Changes in the chemistry of other atmospheric materials can cause a direct effect upon a lake or its watershed without affecting mankind in a global sense. Elements of concern include atmospheric acidity, (e.g., pH), the major constituents which create this acidity (e.g., CO_2, SO_4^{2-} NO_3^-, major constituents which decrease this acidity (e.g., exchange reactions between H^+ and atmospheric clays or silts), and nutrients (e.g., PO_4^{3-}, NO_3, NH_4). The composition of these constituents and projections of their future behaviour is essential for assessment of the long-term background concentrations in lake systems, as these atmospheric constituents strongly influence processes which control these lake concentrations.

Predictions of future atmospheric chemistry will be quite difficult as it will be influenced in unforeseen ways be man acting as a global geochemical agent (e.g., agriculture, the effects polluted river water changes. For changes in acidity, for example, one might consider general scenarios in which the sources of acidity are considered and their input rates to the atmosphere are bounded by estimating future fuel consumption patterns. Fuel consumption of fossil fuels will be constrained by the maximum amount of fuels presently stored in the ground (coal, gas, oil). Proven reserves and probably reserves are useful constraints; the probable maximum amount of fossil fuels formed over geological time in certain types of sedimentary basins may be an alternative constraint.

6.1.6 Prediction of Climatic Change

Possible Measurements

1. knowledge of controls on past climatic trends
2. reliability of future predictions

The earth's climate changes continuously with time and location. The prediction of future climatic changes is aided directly by using past instrumentally-recorded information and indirectly by using natural historical records such as tree rings, "varved" lake sediments, ice cores from glaciers and the general trend of polar ice movement patterns (Kilborn 1980)*.

Evidence suggests that climatic cycles occur regularly, varying from relatively minor 30-year cycles to major glacial and interglacial cycles in terms of tens to hundreds of thousands of years. Some general characterisitcs of the past climatic records are as follows:

1. The last 10 000 years are abnormal in the sense that such interglacial climates represent only about 10 percent of the climate of the last 1 000 000 years.

2. The last postglacial thermal maximum was reached about 6 000 years ago, with a gradual cooling since then with minor periods of marked cooling followed by temperature recovery such as just ended in the 1940's.

3. Climatic changes during the past 20 000 years are as severe as have occurred in the past 1 000 000 years.

4. Interglacial ages each lasting 10 000 $\pm$ 2 000 years and each followed by a glacial maximum have occurred on the average every 100 000 years during at least the past 500 000 years.

5. About 65 000 000 years ago global climates were substantially warmer than today, and subsequent events may be viewed as part of a very long-period cooling trend.

The emerging picture is one of 100 000 year glacial periods with 10 000 year interglacial periods. If we were to presume a continuance of the glaciation events, the next glaciation could possibly begin with a few thousand years although there is a finite probability that a serious worldwide cooling could occur within the next 100 years. During this period, there would conceivably be an increase in precipitation within the northern hemisphere and with the polar ice cap buildup a drop in ocean levels.

* This sub-section is based to a large extent on the report of Kilborn (1980)

The prediction of climate is clearly an enormously complex problem and is influenced by natural cyclic phenomena as well as by such factors as:

1. changes induced by aerosols added to the atmosphere by volcanic eruptions and man-made technology,

2. increased levels of carbon dioxide in the atmosphere due to fossil fuel oxidation.

3. future attempts by man to control climate on a global scale.

6.1.7 Perspective on Atmospheric Characteristics

The long-term character of the atmosphere will strongly influence a lake system. It provides the physical energy needed for significantly disturbing tailings placed in lake sediments systems.

It is suggested that future atmospheric characteristics be viewed from a point of view of whether they have mainly a regional impact or a global impact. If the impact is global, then potential tailings mobilization must be viewed in a sense relative to the dangers to man caused by other changes in the global ecosystem. For example, if the CO_2 concentration or the heat content of the atmospheric changes, tailings may be increasingly mobilizable; however, the effects of changes in heat content directly or indirectly on mans' well-being may be judged to be more substantial (e.g., due to changes in the ability of land to produce food) than mobilization of all or part of the heavy metals and radioisotopes found in the tailings. This suggests the need for use of risk assessment to bound the extent to which long-term characteristics of relatively unpredictable atmospheric characterisitcs are relevant for consideration as criteria for tailings management in lake systems.

6.2 ATMOSPHERIC INTERACTIONS WITH DRAINAGE BASIN

In this section, those aspects of atmospheric properties which interact with and modify properties of a drainage basin are considered. Attempts are made to differentiate these properties from actual atmospherically induced interactions between the drainage basin and the lake. This distinction is attempted because the same atmospherically induced interactions also are paralleled by direct atmosphere-lake interactions. The differentiation is for the most part successful, but partial overlap between this section and section 6.4 exists and is unavoidable.

Atmospheric interactions with the drainage basin include exchanges of water (hydrologic budget), exchanges of matter (material budget) and perturbations on the physical and biochemical character of the watershed caused by atmospheric extrema. These are now considered.

6.2.1 Hydrological Budget

Possible Measurements

1. average annual rainfall rate
2. average annual evapotranspiration
3. intensity-duration-frequency curves for rainfall
4. average annual runoff
5. statistics of runoff

The major additions of water to a drainage basin generally come from rainfall; in some mountain areas, dew and water vapor leached from passing air by tree needles are significant. Major losses are groundwater infiltration, surface runoff and evapotranspiration. Rainfall is generally episodic (a distinct event) and arises from air masses laden with water moving across the drainage basin. Hence, its temporal pattern is quite variable on a short-term and on a long-term basis. The drainage basin acts to store temporarily a portion of the rainfall, delaying the timing and decreasing the amount of runoff. Runoff patterns are much less variable than rainfall patterns in a relative sense. Rainfall can pass directly into groundwater through the soil or marshes if the rainfall rate is less than the maximum rate of infiltration; otherwise, surface storage in the soil and upper ground water is the source of the slower, sustained rates of groundwater flow and evapotranspiration.

Measurements of the various water fluxes taken together are called the hydrological budget. Measurements of the variability of each flux are called the statistics of rainfall, of runoff, etc.

The hydrological budget of a drainage basin describes the quantity of rainfall which falls on a drainage basin (e.g., mm/year), the associated loss mechanisms from the drainage basin (groundwater flow, m^3/yr., surface runoff, m^3/yr., evapotranspiration, mm/yr.) and storage (e.g., soil moisture).

The "intensity-duration-frequency curves for rainfall and statistics of runoff" describe the frequency of occurrence of different magnitudes of events. For rainfall, two examples are i) an intensity of 0.05 m/hr occurring for a duration of 10 min. with a frequency of once every 5 years, and ii) an intensity of 0.96 m/hr. occurring for a duration of 8 hr. with a frequency once every 100 years; the latter is a simplification of the design storm for southern Ontario, based upon Hurricane Hazel. For runoff, two examples are a 7 day-100 year low flow (a 7 day average low flow which occurs once every 100 yr.) or a 1-100 yr. flood flow (that flow rate, in cfs, which is expected to occur once every 100 yr. on the long-term average). Runoff is further considered in section 6.4.

The statistics of rainfall together with knowledge of other perturbations on the watershed permit inferences about the long-term likelihood of perturbations in export of materials from the watershed to the lake.

6.2.2 Material Budget

Possible Measurements

1. quantity of elements deposited on drainage basin (g/ha-yr.)
2. quantity of material removed from drainage basin (g/ha-yr.)

Rainfall is not a pure substance, but rather contains many impurities including dissolved gases (e.g., CO_2), atmospheric particulates, (e.g., suspended solids), elements emitted to the atmosphere by industrial society (e.g., SO_2, NO_x) and elements which have been transformed by atmospheric processes (e.g., H^+, SO^{2-}, NO_3). Typical concentrations and deposition rates from the atmosphere are given in Table 6.2 (Likens et al, 1977).

Table 6.2: Rainfall chemistry and watershed budget for Hubbard Brook (after Likens et al. 1977)

Substance	Precipitation, mg/l			
	$\bar{x}$	$s_{\bar{x}}$	n^a	$s_{\bar{x}}/\bar{x}$, %
Ca^{2+}	0.17	0.02	11	11.8
Mg^{2+}	0.05	0.01	11	20.0
K^+	0.07	0.02	11	28.6
Na^+	0.12	0.01	11	8.3
Al^{3+}	—	—	—	—
NH_4^+	0.22	0.01	10	4.5
H^+	0.0739	0.003	10	4.4
SO_4^{2-}	2.87	0.11	10	3.8
NO_3^-	1.43	0.11	10	7.7
Cl^-	0.51	0.09	7	17.6
PO_4^{3-}	0.008	—	2	—
HCO_3^-	—	—	—	—
Dissolved silica	—	—	—	—

[a]years of measurement.

Component	Chemical Element							
	Ca	Mg	Na	K	N	S	P	Cl
	Standing stock (kg/ha)							
Aboveground biomass	383	36	1.6	155	351	42	34	•
Belowground biomass	101	13	3.8	63	181	17	53	•
Forest floor	372	38	3.6	66	1256	124	78	•
	Annual flux (kg/ha·yr)							
Bulk precipitation input	2.2	0.6	1.6	0.9	6.5	12.7	0.04	6.2
Gaseous or aerosol input	•	•	•	•	14.2	6.1	•	?
Weathering release	21.1	3.5	5.8	7.1	0	0.8	?	•
Streamwater output								
Dissolved substances	13.7	3.1	7.2	1.9	3.9	17.6	0.01	4.6
Particulate matter	0.2	0.2	0.2	0.5	0.1	<0.1	0.01	•
Vegetation uptake	62.2	9.3	34.8	64.3	79.6[a]	24.5[a]	8.9	•
Litter fall	40.7	5.9	0.1	18.3	54.2	5.8	4.0	•
Root litter	3.2	0.5	0.01	2.1	6.2	0.6	1.7	•
Throughfall and stemflow	6.7	2.0	0.3	30.1	9.3	21.0	0.7	4.4
Root exudates	3.5	0.2	34.2	8.0	0.9	1.9	0.2	1.8
Net mineralization	42.4	6.1	0.1	20.1	69.6	5.7	?	?
Aboveground biomass accretion	5.4	0.4	0.03	4.3	4.8	0.8	0.9	•
Belowground biomass accretion	2.7	0.3	0.12	1.5	4.2	0.4	1.4	•
Forest floor accretion	1.4	0.2	0.02	0.3	7.7	0.8	0.5	•

These materials represent a source of essential nutrients (e.g., NO_3) for plants and are a driving force for the weathering reactions of soils and bedrock due to the natural acidity of rainwater. Atmospheric long-range transport and deposition are thought to be the main sources of organic pesticides in areas not sprayed.

Removal from a drainage basin to the atmosphere is quite significant for many elements. Wind driven erosion of non-vegetated areas can pick up soil particles and carry them long distances. Even in many urban centres, soil particles represent a large fraction of the total atmospheric concentration of particles. Radioisotopes associated with particles can be a significant flux in dry areas (e.g., New Mexico) or on dry tailings piles (Kilborn, 1980). Soil and marsh decay is a significant source for the efflux of such elements as methane to the atmosphere; radon degassing from exposed deposits of radioactive elements can also be significant.

Few data are available on the deposition to or removal of radionuclides from drainage basins, except for specific locations. It is generally a local problem near an exposed bedrock formation or a human activity such as mining.

(Note, export from the watershed through aquatic pathways is considered in Section 5.2 and in 6.4).

6.2.3 Atmospheric Perturbations on Drainage Basin

Possible Measurement

1. history of droughts
2. history of flash-floods
3. history of natural forest fires
4. estimates of future rainfall chemistry

The main perturbations of concern here are those atmospheric events which significantly modify the character of the watershed. A prolonged drought will cause a shift in the distribution of trees, grasses and other plants. High winds associated with tornadoes, hurricanes, and large rainstorms will uproot trees. The possibility of such damage might be assessed from characteristics of floods or of winds on the region. Natural forest fires are caused by lightning; these fires may completely remove the vegetative cover.

In the short-term, such perturbations as droughts, high winds and forest fires significantly alter the watershed ecosystem. However, in the long-term, it is a reasonable hypothesis that a watershed ecosystem is adapted to such events and will self-regenerate within a period of time. This does not, however, imply that the watershed ecosystem is adapted to a long-term shift in climate (e.g., towards wetter or dryer climatic conditions) or to forest fires. Many forest fires, presently occurring, are caused by mankind.

Perhaps the largest perceivable long-term perturbation is a sustained shift in rainfall chemistry. Increased acidity will probably cause a significant shift in the vegetative cover of watersheds and in soil characteristics in ways which are not presently foreseen.

6.2.4 Perspective on Atmospheric Interactions with Drainage Basin

Atmospheric interactions with a drainage basin are, in general, not a significant pathway for movement of radioisotopes, except in local areas. These interactions are mainly significant in the effects they have on the stability of the watershed ecosystem, the internal material fluxes of the ecosystem, and the export of materials (water, elements) from the watershed to the lake.

6.3 ATMOSPHERIC INTERACTIONS WITH LAKE

In this section, the influence that the atmosphere has on the lake's physical and chemical properties are examined; namely, exchanges of energy (heat and mechanical) and of mass. Primary emphasis is placed upon various conceptual approaches to relating the atmospheric exchange of energy with lakes to the properties of the atmosphere. Due to the overlap between this section and 6.4 with respect to exchanges of mass (water, chemistry) and their event-oriented properties, in-depth coverage is provided in section 6.4.

In this section, exchanges of heat are described under the titles of lake heat budget and solar radiation-lake interactions; exchanges of mechanical energy are examined in the sub-section on wind shear stress-lake current relationships. The term wind shear stress is used as it is a concept used by various branches of engineering to describe, quantitatively, the magnitude of a horizontal force "felt" by the surface as a wind moves parallel to and across the surface. It is a measure of the friction exerted on the water by the wind.

6.3.1 Lake Heat Budget

Possible Measurements

1. annual cycle of equilibrium temperature
2. various heat fluxes

The lake heat budget is the qualitative (and quantitative) description of the major mechanisms of exchange of heat between the lake, and its environs, namely the atmosphere and the earth's crust. The lake heat budget is dominated by exchange with the atmosphere; exchange with land in the form of river flows, groundwater flows and heat conduction are rarely important compared to the atmospheric exchange and are normally ignored. The lake heat budget is normally described as being composed only of mechanisms which exchange heat with the atmosphere.

The equilibrium temperature of a lake is the surface water temperature at equilibirum with the atmosphere. It is defined as the hypothetical water surface temperature at which the net rate of surface heat exchange would be zero (Edinger et al, 1974). It provides a quantitative measure of the expected water temperature, if the downward

transport of heat into deeper epilimnetic waters, the conduction of heat to lake sediments, and the effects of flowing water are negligible. Since average monthly temperatures of many lakes do approach average equilibrium temperatures, the equilibrium temperature is one method for estimating the surface thermal characteristics of lakes where measurements are not available.

To calculate the equilibrium temperature, knowledge of five main heat fluxes between the atmosphere and lake are needed. Solar radiation (short-wave) and atmospheric radiation (long-wave) are the two main heat transfer mechanisms to the lake; back-radiation (lake to atmosphere, long-wave), convection and evaporation are the three main loss mechanisms. Solar radiation may be measured (e.g., with an Eppley Pyrhelimometer) or calculated (as a function of latitude, time of year, clarity of the sky, cloudiness, etc.). The other four fluxes may be calculated with sufficient accuracy using various formulae. Atmospheric radiation is a function of absolute air temperature, relative humidity, and cloud type and cover. Back-radiation is a function of the emissivity of the water surface and absolute water temperature. Evaporation is a function of the windspeed, the air temperature, the water temperature and the relative humidity. Convection is a function of evaporation. Of the three loss mechanisms, knowledge about convection is the least developed experimentally, but its magnitude is smallest. The absolute error associated with convection, hence, is small.

If there is a significant error in the heat budget, estimation of evaporation is usually the main source (Tetzlaff, 1979; Octavio et al, 1976). The main error is in the wind speed part of the formulation and in using tower data, taken at a 10 m height on land, as a measure of wind speed over the lake surface. Use of the wind speed data is usually the main error because of the variations in the effective thickness of the "atmosphereic boundary layer" (see section 6.4). In one case, the approach taken was to reduce the wind speed recorded on a tower by up to 50%; this reduction makes the evaporation estimates fairly precise.

The lake heat budget has three particular uses. First, it allows calculation of the equilibrium temperature. This is important for predicting biological regimes. Second, it is a check on evaporation formulae; for lakes in which land-based inflows are negligible, evaporation may be greater than, equal to or less than rainfall.

Relatively precise knowledge of evaporation permits statements about whether or not the lake will be an overflowing lake in the long-term (e.g., compare an arid climate to a wet sea climate). Third, it permits estimates of the length of time of ice cover. Such information is needed for mathematical models which make predictions of vertical temperature regimes and of vertical circulation patterns in natural or artificially chemically stratified lakes.

6.3.2 Solar Radiation - Lake Interactions

Possible Measurements

1. percentage of short-wave radiation adsorbed at lake surface
2. net extinction coefficient
3. compensation depth

Clear, deep lakes are occasionally observed not to stratify thermally while nearby shallower, less clear lakes, do stratify during the summertime. This occurs even under similar atmospheric characteristics (solar and atmospheric radiation, wind speed at the lake surface, relative humidity). The main cause of these differences is the depth of penetration of sunlight.

For many waters, approximately 50% of the total energy of solar radiation (units $Kcal/m^2$-day) is absorbed in the top few centimeters while the remainder is transmitted through to deeper waters. The split between surface absorbance and surface transmission is fairly constant but may vary from lake to lake and with latitude. In most lakes, wavelengths in the red part of the light spectrum are nearly totally absorbed, while wavelengths in the opposite end of the spectrum (blue-violet and shorter) are almost completely transmitted. In humic (brown-water) lakes, the opposite partially occurs - blue wavelengths are absorbed by humic material while red wavelengths penetrate more deeply. Brown lakes are typical of many shield lakes. If atmospheric adsorption occurs as a function of location (latitude) or time of year, such that differing quantities of red wavelengths are incident upon the lake, the percentage absorbance of total sunlight energy will vary.

The net logarithmic extinction coefficient is a measure of the amount of incident surface light absorbed or scattered by a given depth of water. It is a function of light absorption by water, absorption and scattering by suspended particles and absorption by dissolved components (e.g., colour). Higher concentrations of algae, etc. increase the extinction coefficient. For values of 1-2 m^{-1},

typical of turbid lakes, 99% of transmitted light is absorbed within the top 2-4 m, while for a value of 0.1 m^{-1} typical of quite clear lakes, 99% of the transmitted light is not absorbed until a depth of 46 m. This difference is sufficient to prevent the formation of summer stratification in deep Canadian lakes. Because lakes in more northern regions have lower surface equilibrium temperatures, higher values than 0.1 m^{-1} are still sufficient to prevent formation of thermal stratification.

6.3.3 Wind Shear Stress - Lake Current Relationships

Measurements

1. wind shear stress
2. empirical wind velocity - lake current formulae

Perhaps one of the most difficult aspects of predicting or estimating lake physics (horizontal and vertical characteristics) is the estimation of the effects of winds. Such effects include horizontal characteristics such as the relationship between currents and winds and vertical characteristics such as the relationship between evaporation and wind velocity or between thermocline entrainment rate and wind velocity.

Most research and empirical efforts try to treat properties of the lake as separate from those of the atmosphere or vice versa. They are, in fact, highly coupled, particularly for large lakes. For example, consideration of Lake Erie subjected to a westerly wind shows that to predict the effects of winds upon the east end of Lake Erie while ignoring the effects of the lake upon the winds in the west end is difficult. Further, to ignore the effect of the Great Lakes upon the regional climate (cooler in summer, warmer in winter than similar land masses) is erroneous. Particularly in ocean work, but also in lake work, Priestley and Taylor (1972) made a compelling plea for treating the atmospheric and surface water regimes as coupled rather than separately as is currently the normal approach for most lakes. They assert that one of the main limitations to current techniques is this separation.

But with this in mind, we will consider them as independant entitles assuming that atmospheric processes drive lake processes. This is the current state of the art and is quite a useful approximation for our purposes; it is probably limited, in the strictest sense, to relatively small lakes but is useful for larger lakes.

Lake currents may be estimated as being equal to the wind speed times the wind shear stress (i.e., the wind drag). One approach is to use available wind data and to relate the wind shear stress to such macro-lake properties as fetch (the longest length of the lake in the direction of the wind). Another is to change the recorded wind speed in some known manner to reflect atmospheric properties and to use a wind shear stress formulation which is a function only of properties of the water waves (and which is independent of bulk lake properties).

In a capsular sense, the key problem is "how does one use wind data from a land-based meteorological tower taken at a height of 10 m to estimate the effective wind velocity at a height of 2 m (or some other reference height) directly above water? This is an atmospheric "boundary layer problem", i.e., a problem which considers that part of the atmosphere which is in contact with the earth, typically a thickness of 0.01-10 m. To facilitate this calculation properly, it would be desirable to use a mathematical model of atmospheric physics. Another alternative is simply to change (increase or decrease) measured wind speeds by a needed amount to get a good fit with observed data (e.g., the effective wind speed for evaporation formula is reduced to obtain a proper correspondence with observed evaporation rates).

These alternatives for calculating lake currents as a function of wind characteristics devolve into two approaches, a micro- and a macro-. The micro approach attempts to relate a lake surface current directly to a given wind speed using the concept of wind shear stress. A macro-approach relates internal lake properties to wind speed directly without calculating surface currents; the macro-approach implicitly considers surface currents since it is based upon the concept of wind drag. The micro-approach is particularly useful for horizontal currents while the macro-approach is particularly useful for vertical properties (e.g., thermocline entrainment rate). The state of the art uses relatively simple relationships (partly empirical, partly theoretical) rather than elaborate models of the physics of the atmospheric-lake boundary layer.

The wind shear stress is a function of the wind speed, wave height, spectrum of waves, and perhaps fetch. A simplifying hypothesis widely used is that the efficiency of wind energy transfer to lake kinetic energy is dependent solely upon small, high-frequency waves which ride the top of the big waves, rather than upon the big waves. Such small waves are independent of the fetch. Further, due to the variability of

natural winds, true fetch-limited conditions for large waves are not obtained and hence, the fetch does not control large wave development in large lakes (Plate and Wengefeld, 1979). These points imply that if one has measurements of wind velocity directly above the lake surface, calculation of the transfer of kinetic energy to lake water can be made in a fairly straight-forward fashion. Hence, the effects of fetch observed in the literature upon transfer of kinetic energy are due more to variations in properties of the atmosphere than to variations in the physics of surface lake currents.

The effects of variations in the "thickness" or "height" of the atmospheric boundary layer is shown in the next few sketches. An offshore wind (Figure 6.3.1) with a velocity profile reflecting land effects, gradually adapts to the water surface properties. The figure shows the feedback loop between air flow and water flow through the effects of water surface shear stress and water wave generation. It induces a surface current in the offshore direction and a bottom current in the onshore direction. Abstraction of the offshore wind (Figure 6.3.2) into a model geometry permitted them to derive a simple analytical expression describing current velocities until approximate constant conditions (Region IV) are obtained.

The weakness to such an approach is that lake shore lines are not regularly shaped, but rather quite irregular. The concept behind the lee-zone effects of Hansen (1979) is shown in Figure 6.3.3. His findings for a small lake (Esrom, Denmark) are quite relevant here. The lake is small (17 km^2 surface area). It is surrounded by forest-covered hills, 30-80 m above the lake surface; the trees are 10-20 m high. Much of the lake is in the lee of the hills. He estimates that the wind shear stress will be zero for a length equal to six times the height of surrounding hills and that a further distance of seven times the height is required before surface currents reach their full value. For this small lake (main axis is north-south, predominant winds are westerly), Hansen estimated that over one-half of the lake surface is influenced by the lee-zone effects. This is the reason for the need to reduce the effective wind-speed for evaporation calculations. The lee-effects, and development of the atmospheric boundary layer are the main reasons why correlations have been found between wind shear stress characteristics of a lake and its fetch.

Methods for using a macro-approach for relating lake currents to winds are now examined. These methods calculate the entrainment rate at any point in time as a function of dimensionless numbers. To make these calculations over time, use is made of temperature models.

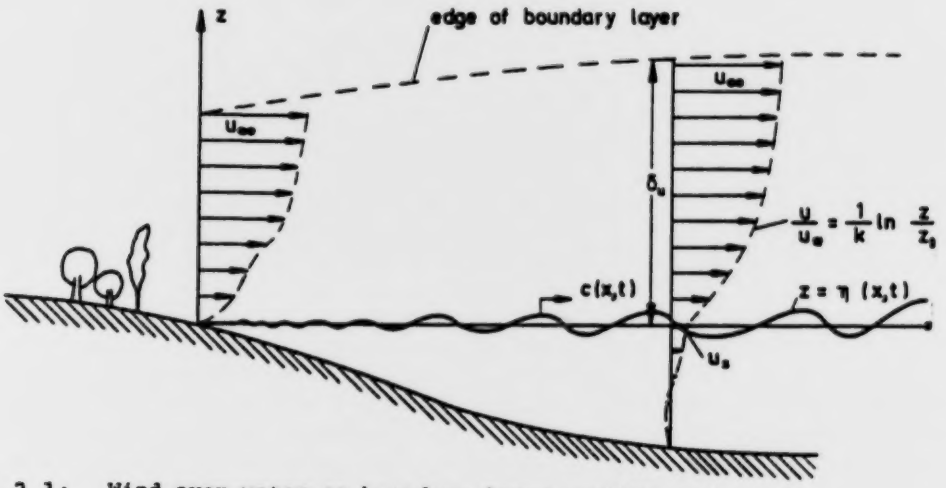
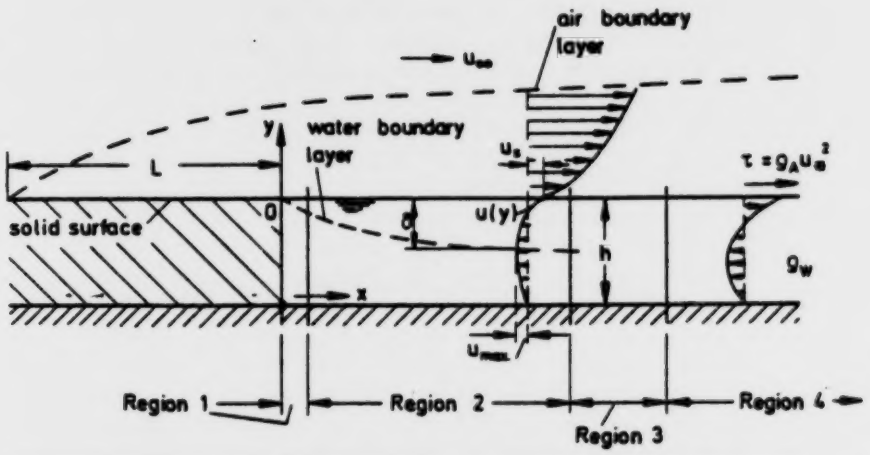


Figure 6.3.1: Wind over water as boundary layer problem

(after Plate and Wengefield, 1979)



		Region 2	Region 4
$\frac{u_s}{u_*} =$	laminar flow	$1.134 \left(\frac{\varrho_A}{\varrho_w}\right)^{2/3} \left(\frac{u_* x}{\nu_w}\right)^{1/3}$	$0.25 \frac{\varrho_A}{\varrho_w} \cdot \frac{u_* h}{\nu_w}$
	turbulent flow	—	$0.026 \frac{1}{\sqrt{C_0}}$

Figure 6.3.2: Regions of water surface drift velocity and results

(after Plate and Wengefield, 1979)

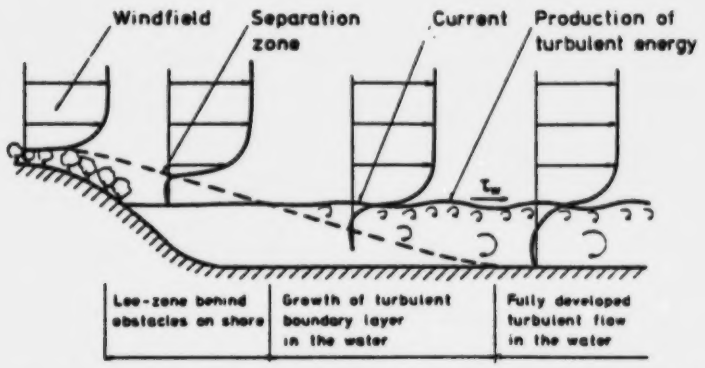


Figure 6.3.3: The development of a Flow Field in a Lake. τ_w is the wind shear stress on the surface of the lake.

(after Hansen, 1979)

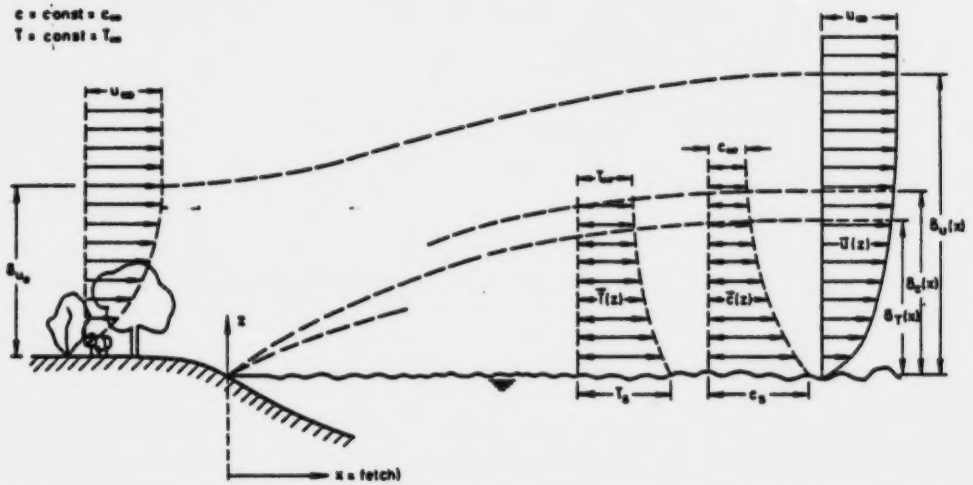


Figure 6.3.4: Definition-sketch for the boundary layer problem of momentum-, heat- and mass transfer at a water surface

(after Plate and Wengefield, 1979)

Vertical thermocline entrainment rates are frequently calculated as a function of wind velocity and of internal lake properties such as the Richardson number. Octavio et al (1976), Hanson (1979) and others have related the entrainment rate to the wind velocity and a "Bulk" Richardson Number (BRN) which differs from the gradient Richardson Number given in the Lake Physics section. The BRN is defined as

$$BRN = \frac{g(d\rho/dz)\, h^2}{2\, \rho_o\, u^2}$$

where h is the depth of the mixed layer, p is the water density, u is a surface shear velocity and is a function of the wind speed. The Bulk Richardson number is a measure of total water column stability (e.g., over the epilimnetic depth) with respect to surface current properties while the densimetric Richardson Number is a measure of stability between adjacent waters within the water column.

Various investigators (e.g., Octavio et al, 1976) have made predictions of the entrainment rate of the thermocline using these entrainment rate formulations and a temperature model. On a daily basis and for given wind conditions, they have had good success at making these predictions. The temperature model is needed in order to specify density variations over time and the response of the density to the interactions of surface heating and wind driven mixing. A typical temperature model includes the major heat fluxes across the lake surface, the effects of inflowing and outflowing waters and vertical entrainment. Such a model provided the basis for BEAK's work aimed at examining the relationships of wind driven mixing in lakes with artificial chemical stratification.

These daily temperature models work quite well ($\pm$ 1°C) for the upper waters of lakes such as Canadian Shield Lakes, where wind data are available from a meteorological tower adjacent to the lake and where lee problems are not overly extensive. Where lee problems are extensive, use of Hanson's (1979) relationships for mixing rates as a function of Richardson Number, sheltering effects and fetch length-to-depth ratio are useful. Where wind tower data are available, some distance away and separated from the lake by hills or mountains, atmospheric boundary effects need to be considered. Linear reduction of wind speed for evaporation calculations is one approach. Figure 6.3.4 shows the effective thickness of boundary effects for transfer of

different atmospheric properties. It suggests that the reduction factor for wind speed (momentum; height $S_u(x)$, velocity u) will be different than for temperature (heat; height $S_T(x)$, temperature T) and for water vapour (mass; height $S_c(x)$; concentration C).

For use of such temperature models, one significant criterion is that the thermocline should not tilt so much that the hypolimnion would reach the surface. As these models do not consider "leakage around the side", they will over-predict the development of stratification.

Such leakage around the side is typical of large lakes subject to strong winds. A useful criteria is that the thermocline depth Ze should be such that

$$Ze \leq \frac{4 \times 10\text{-}7w2}{z} \frac{\rho}{\Delta\rho} \frac{L_t}{2}$$

Here z is the mean water depth (m), $\Delta\rho/\rho$ is the relative density difference between epilimnion and hypolimnion and L_t is the mean lake length in the direction of the wind. W, the wind speed (m/s) is the maximum wind speed.

6.3.4 Water Budget and Mass Budget

Possible Measurements

1. ratio of rainfall to evaporation
2. atmospheric deposition rate of particulates
3. atmospheric deposition rate of chemical constituents

Atmospheric deposition of rainfall, particulates and chemical constituents of rainfall can be a significant portion of the total inputs of mass to a lake. These processes are especially important in headwater lakes.

The ratio of rainfall to evaporation gives an indication of the long-term tendency of the lake level if no drainage basin inputs to the lake were allowed. It gives a preliminary indication of the utility of a mitigating measure such as bypassing all inflow streams around the lake.

As described earlier, fairly precise estimates of each may be made using measurements of atmospheric properties made by AES and suitable formulae.

The atmospheric deposition of chemical constituents of rainfall and of particulates quantifies that portion of a lake's material budget which comes from the atmosphere. If existing land-based inflows to a lake were bypassed around the lake, it would become the dominant long-term input of mass to many lakes. Several networks now exist for measuring such deposition; they vary from being quite dense in Ontario to being quite sparse in northern areas. Because they have only existed for 5 to 10 years, extrapolation to the long term is difficult.

6.3.5 Perspective on Atmospheric - Lake Interactions

A heat budget provides an excellent tool for estimating the surface water temperatures of a lake caused by long-term climatic variations and for estimating long-term evaporation by checking the evaporation formula. Surface temperatures are useful for estimating the length of the ice-free season and for estimating probable biological regimes. Evaporation estimates permit statements about the relative importance of evaporation to inflow, outflow, and precipitation for controlling lake levels. Evaporation formulae are, for practical purposes, as precise as measurements (Neuwirth, 1979).

Knowledge of the extinction coefficient is essential for determining whether or not a lake will stratify. It is a function of water colour, turbidity and algal biomass.

Wind shear stress characteristics need to be estimated for predicting horizontal lake currents, and vertical circulation characteristics. They, together with vertical temperature models, provide an ability for predicting vertical lake temperatures with a high degree of accuracy. Such information is required where artificial stratification of lakes by chemical means is being contemplated in order to predict the rate of breakdown of the chemostratification. It is needed for predicting weekly surface temperatures on a fairly precise basis. Such information is not needed if the smallest time scale of concern is one year and if the lake mixes vertically at least once per year such that it approaches well-mixed conditions.

Estimation of wind shear stress characteristics is a rapidly evolving field. It has at least two limitations. Present methods for transforming wind (and other meteorological) data from a 10 m tower several km away to represent wind characteristics above a lake need further development. The consideration of lee effects in small lakes needs further developmental work.

Water and chemical inputs from the atmosphere to a lake may be a substantial portion of the lake's hydrological and material budget. Their importance for determining long-term background concentrations of water constituents (e.g., nutrients, radionuclides) depends upon their relative proportions. Long-term predictions of rainfall will depend upon our ability to make predictions of long-term changes in climatic regimes; long-term predictions of rainfall chemistry is much more difficult due to the lack of adequate measurements over a wide area for a long time period, and due to a lack of understanding of all of the relationships which determine atmospheric chemistry.

6.4 ATMOSPHERIC INTERACTIONS WITH DRAINAGE BASIN - LAKE SYSTEM

6.4.1 Geochemical Weathering Reactions (see Figure 6.4.1)

Possible Measurements

1. watershed material budgets
2. shifts in watershed material budgets

Comparison of atmospheric deposition (dryfall, rainfall, interception, impaction) to watershed export via surface runoff gives a quantitative measure of the dominant geochemical weathering reactions for a watershed. This comparison assumes that the amount of material which leaves a watershed via a groundwater pathway and via an atmospheric pathway is minimal. The difference between atmospheric deposition and watershed export is not completely due to geochemical weathering reactions but rather is the sum of weathering reactions and their modification by elemental cycles between the watershed vegetation and watershed soils.

A typical watershed budget is shown for an Appalachian Mountain watershed in Table 6.2. It has a mixed deciduous-coniferous forest covering bedrock and shallow tills; watershed slopes vary from steep in the upper reaches to shallow in the lower reaches (Likens et al, 1977). Overall, more cations (positively charged ions, attracted to a cathode) and anions (negatively charged ions, attracted to an anode) leave the watershed than are deposited in it. Since the electroneutrality balance is not maintained on an input-output basis, the difference represents the degree of liberation of ions by weathering reactions, by leaching, and by ecosystem mobilization from the watershed's soils and bedrock (e.g., removal from soil by vegetation and eventual release to soil or water). In particular, acids (hydrogen ions) are absorbed and cations such as calcium, magnesium, sodium and potassium are released.

The pH of pure water at equilibrium with the atmosphere is 5.7 while the pH of typical rain water falling on Eastern Canada and B.C. varies from 3.5-5. The pH of typical rainwater on the praries is greater than 6. The pH of many lakes on the Canadian Shield varies from approximately 6.5-6.8 if they are not subjected to "acid rainfall", to above 6.8 if the watershed has substantial calcarious materials and to below this range depending upon the character of acid rain and the

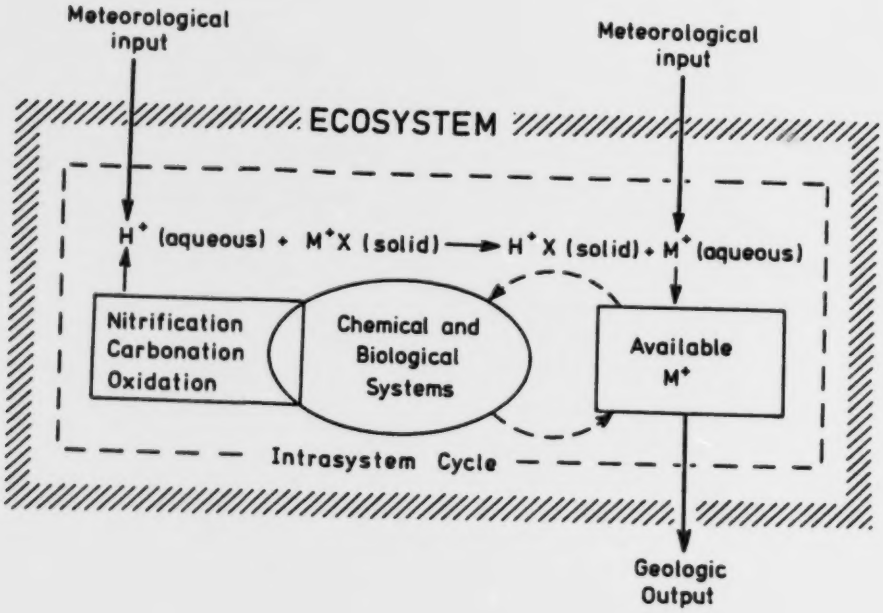


Figure 6.4.1: Description of a watershed's weathering reaction

M^+ represents a basic cation and X is an ionic exchange substrate, usually a primary silicate mineral, but also includes such derivatives as soil minerals or soil humates. The above illustrates that as H^+ is added to the system from external and internal sources, basic cations (M^+) are leached from the system, and primary silicate minerals (M^+X) are transformed into secondary minerals (H^+X).

buffering effects of watersheds. If the natural pH of rainfall is in the range of 5.3-5.7, hydrogen ion concentrations decrease as the water traverses the watershed from rainwater to lakewater. This decrease provides an indication of the natural absorption capacity of many watersheds. Deviations from such relationships are measures of the degree of shift of watershed budgets for lakes on the Precambrian Shield.

Watershed budgets are presently available for many areas of Canada, the United States, Western Europe, Iron Curtain countries and tropical countries. The majority are for agricultural areas and "disturbed" watersheds, but there are substantial numbers of studies which have been conducted upon watersheds similar to those of the Canadian Shield and Rocky Mountains. Estimates of export from an unmeasured watershed can be made with some confidence using data for similar watersheds. Such estimates should be made before a measurement program is initiated. A more reliable method is to measure the watershed export. This, coupled with estimates or measurements of atmospheric deposition, allow estimation of the present watershed weathering reactions. Such a comparison is in a sense a "calibration" of a land runoff-export model.

For the aim of estimating long-term shifts in geochemical weathering reactions, projections of long-term shifts in atmospheric characteristics (rainfall and its chemistry) are required. Selection of data from watersheds whose atmospheric chemistry and rainfall are similar to those of the lake under consideration for tailings disposal will provide a basis for estimating the long-term shift of watershed weathering reactions. However, caution is in order since these watershed budgets are relatively coarse and since ecosystem processes may shift more rapidly or less rapidly than literature data suggest. The watershed budget should be used more to indicate gross tendencies of watershed export rather than to make precise predictions.

6.4.2 Long-term Trends in Lake Chemistry

Possible Measurements

1. ratio of drainage basin area to lake surface area
2. change in watershed material budgets

In general, a lake is a net depositional environment. That is, less material leaves the lake through the surface outflow and through the groundwater outflow than enters through surface inflows and

groundwater inflows. A few materials (e.g., chlorides) are conservative (long-term outflow = long-term inflow). Only for a very few materials, are sediment sources a significant input to the lake water (compared to surface water inputs). Such materials include those which may leach from deposits placed in the lake environment by man. Hence, in general, the lake concentration is "driven by" or is "controlled by" the input rate of materials from the watershed and from the atmosphere.

For headwater lakes, direct input of different elements from the atmosphere to the lake may be equal to inputs to the lake from the drainage basin. Such materials include SO_4^{2-}, H^+ and CO_2. For many lakes (particularly where the drainage basin area is substantially greater than the lake surface area), inputs to the lake from the watershed are much larger than inputs from the atmosphere. The ratio of drainage basin area to lake surface area provides an estimate of the dominant cause of any future shifts in lake water chemistry.

Shifts in material budgets, whether due to shifts in watershed export or due to shifts in atmospheric deposition, will be a dominant cause of changes in water chemistry. This does not minimize the importance of future changes in settling rates of lake particulates or changes in the behaviour of uptake relationships between lake soluble constituents and particulates. Rather, it states that a prime determinant will be a shift in amounts of material entering the water column.

6.4.3 Erosional Processes

Possible Measurements

1. suspended solids and bedload export from watershed
2. suspended solids - flow rate relationships
3. rate of recession of shore-line bluffs
4. sedimentation rates in a lake
5. depth of groundwater relative to the frost line

The amount of erosion from a watershed determines the sedimentary nature of the lake and the rate of infilling. It determines the rate at which tailings will be covered in the future and the nature of the sediments (the organic-inorganic mix) which will be covering the tailings.

Erosion of soils and bedrock is affected by the erosive forces caused by wind and rain, gravitational effects and bedrock stability (e.g., landslides where the terrain is sufficiently steep), and the long-term tilting characteristics of the bedrock. Erosion is catalyzed by freeze-thaw cycles due to the expansion of water in rock and soil pores. Erosion is minimized by vegetative cover on the watershed.

The amount of particulate material that enters a lake's sediment comes from four major sources: river-carried organics and inorganics, overland transport directly to a lake (called sheet-erosion), erosion of shoreline bluffs and atmospheric erosion. River export of particulates occurs via two main transport routes, suspended solids and bedload transport. Sheet erosion in the watershed is directed both toward the rivers and toward the lake depending upon watershed topography. That portion which enters the stream becomes a part of its bedload or suspended solids. Shoreline bluffs are generally eroding features composed of sandy or silty relief with steep slopes. Of the four sources, stream transport and overland transport are universally significant. Shoreline erosion will be significant only where necessary features exist. Atmospheric transport is typically a small quantity relative to the other sources.

To assess the erosional character of watersheds in a wide range of lakes, interpretation of maps and air photos are useful. Soil survey maps provide descriptions of the erosional character of various terrains. Surficial geology maps and knowledge from the field of physiography provide information on types of bedrock and likely fracture characteristics. Inspection of air photographs provide information on erosional features; their historical character and possible interferences by vegetative cover during their photography limits their usefulness to some degree. Mapping available in the Natural Air Photo Library Ottawa and in the Ontario Centre for Remote Sensing is valuable for assessing existing features.

To assess one or several watersheds, more detailed information is useful. Interpretation of daily Cansat photography keyed to the timing of rainfall and high-flow conditions or drought conditions gives qualitative information on erosional patterns. Infrared photography of river-lake areas on a frequent basis provides information concerning the sediment loads of rivers after rainstorms.

To provide precise information on a watershed, estimates may be made using unit area export coefficients for suspended solids; alternatively, actual river measurements may be made for one to two years. Estimates may be made using the approach of the universal soil loss equation (USLE). The USLE calculates export as a product of several factors including watershed slope, length of overland flow, soil characteristics (e.g., texture), vegetative cover and management factors. Specific values for each factor are assigned based upon field measurements of export from many watersheds and statistical analysis. Application to a specific area of Canada generally requires a check on the "value" system for each factor and sometimes assignment of new numbers. Its applicability is also limited because its greatest use has been for unconsolidated tills, while many lakes which may be considered for tailings disposal will have large areas of their drainage basins covered with bedrock or sparse soil cover.

Measurement of watershed export of sediments involves three specific parameters. These are bedload transport, suspended solids transport and direct sheet erosion to the lake. For the large rivers of the world (e.g., the Amazon, the Yukon river) Gibbs (1975) found that bedload transport is insignificant (Snodgrass, 1977). In rivers draining small, steep watersheds, bedload transport of large-grained particles becomes significant. To measure bedload transport, in situ traps which accumulate sediments being carried along the river bottom are inserted and emptied periodically. The annual amount trapped is an estimate of bedload transport. This quantity is often related to annual flow volume.

Suspended solids export is calculated by measuring the suspended solids concentration at frequent intervals over a hydrograph. Several hydrographs are sampled to allow calculation of export. The concentration of suspended solids (organic or inorganic) in a river is generally observed to increase with the rate of flow. The calculation of export must take account of this relationship, or substantial errors will be produced.

The flow-concentration relationship is a measure of the erosional nature of the watershed - a strong relationship is a measure of a high erosional character. A specific flow-concentration relationship may be a general property of a specific watershed. However, there has not been a sufficient synthesis in the field of hydrology to prove this point as different runoff events have somewhat different flow-concentration relationships in the same watershed.

The rate of recession of shoreline bluffs is a measure of the rate of change of shoreline and of the rate of infilling of the lake. For some lakes situated in unconsolidated tills, it is estimated that input from erosion of bluffs is the same order of magnitude as for riverine inputs. For Canadian Shield lakes, recession of shoreline bluffs is typically quite small.

Coring of lake sediments and examining for specific layers (such as varved sediments or pollen layers) or isotope dating permits determination of the historical rate of accumulation of sediments. Examination for pollen allows an estimate of sediment depth because specific pollens are associated with the development of agriculture in North America. For example, development of agriculture in southern Ontario is associated with the year 1850. Examination for changes in algal depositional patterns (e.g., diatom shells) in association with knowledge of logging history permits dating. Some care must be used in measuring sedimentation rates. These techniques assume that the sediments have been deposited and not disturbed since the time of deposition. Biological reworking or mixing of surficial sediments at some point in the past due to wind-induced mixing of bottom waters can significantly affect the profile. Lake sediments in shallow waters may be erosional areas while sediments in deeper waters will be accumulation areas (see section 2.2). Coring at several stations is required in order to estimate the average sedimentation rate in a lake.

The depth of groundwater relative to the frost line is a measure of the probability of future alteration of weathering reactions due to the freeze-thaw cycle and of future change in erosion rates of bedrock. If bedrock containing groundwater is exposed to the freeze-thaw cycle, it will more readily break up and erode than if it does not contain groundwater. Similar influences of erodability exist for unconsolidated soils in relatively steep terrain.

6.4.4 Surface Runoff Characteristics

Possible Measurements

1. characteristics of runoff from watershed
2. characteristics of runoff from lake
3. lake level

The runoff characteristics from the watershed influence lake physics and the transport capacity of rivers. A lake dampens extreme runoff from the watershed. The degree of dampening influences the runoff characteristics of downstream rivers, particularly in the immediate vicinity of the outflow. If the geological structure controlling the level of outflow is easily erodable, such as an unconsolidated till on highly fractured bedrock, the degree of lake dampening strongly influences the effects that runoff generated from rainstorms or hurricanes has on the maintenance of this structure. Variations in long-term rainfall or evaporation will strongly influence long-term amounts of runoff and the long-term lake level.

A description of runoff characteristics is given in Section 5.2. Two main approaches exist for estimating runoff. One is a deterministic approach in which the physics of water moving through a watershed is used to relate rainfall to runoff. Various deterministic models are capable of predicting a given runoff pattern due to rainfall patterns and watershed characteristics (e.g., Stanford Watershed Model, HYMO, SWMM). Their major differences are the time intervals considered (minute, hour, day) and the degree of sophistication used for describing infiltration-groundwater flow mechanisms. The second approach is a statistical approach. The statistical approach describes runoff as a frequency of occurrence of a certain size of runoff. It assumes that over the long-term, the frequency of occurrence of a certain magnitude of runoff is independent of rainfall patterns and that the watershed characteristics remain constant over that period of time.

The strength of statistics for describing runoff is that it allows statements concerning frequency of occurrence. However, it cannot predict when a certain size of event may occur. A 100-year flood may occur in 2 successive years. This does not mean that the statistics are wrong, but rather that these 2 flows occurred twice in a 100-year period, or that 3 more events of this size are expected in a 500-year period.

Two problems arise from the use of statistics. First, statistical statements are based upon short-term records (most gauging stations have only been in place for 50 years or less). If Canada's climate has been quite stable for the past 30 years compared to a more long-term pattern of wider variations, resultant runoff patterns and their statistics will be misleading. Secondly, changes in the watershed character will alter runoff patterns changing the statistics of runoff. Use of statistics should take into account the fact that they cannot be used to estimate changes in future runoff patterns. Only i) with deterministic cause-effect relationships describing the modifying effects of the watershed upon rainfall to produce runoff, and ii) with estimates of future changes in rainfall and watershed characteristics can future statistics of runoff be estimated.

Surface runoff characteristics of an individual event are strongly influenced by initial soil conditions. This is illustrated by Ontario's experience with Hurricane Hazel. The rainfall quantity in itself is only approximately a 1-100 year storm. Its flooding impact arose from the fact that the soils of the watershed were fully saturated by rain which had fallen during the previous 36 hours. The pattern of rainfall for a design storm based upon Hurricane Hazel is given in Table 6.4.2. The 2.9 in. rainfall received during the first 36 hours saturated the soils and subsequent rainfall from the storm centre (hours 40-48) became runoff with insignificant amounts of surface storage or groundwater infiltration.

For much of Canada, rainfall extrema are usually associated with tornados and violent thunderstorms rather than with hurricanes. The relative importance of each is a function of the geographical location.

For lakes, the short-term characteristics of runoff affect two aspects: their mixing characteristics and the temporal behaviour of the lake surface. Mixing effects are described in the next subsection.

If the volumetric dampening capacity of a lake is large, the downstream hydrograph is highly dampened, both with respect to magnitude of peak flow and length of time of runoff. Such dampening greatly reduces the chances for disturbance of an outflow sill. One possible measure of this dampening suggested by BEAK is the potential rise of lake level from a design storm. This potential rise is equal to the total rainfall of the storm (units of in. or mm) times the drainage area

Table 6.4.2: Regional design storm for southern Ontario

Period of Time (Hrs.)	Rainfall (in.)
0-36	2.9
37	0.25
38	0.17
39	0.25
40	0.50
41	0.66
42	0.50
43	0.91
44	0.5
45	0.5
46	2.08
47	1.49
48	0.5

This storm is applied to a drainage basin 0-10 sq. mi. in surface area. If the area is greater than 10 sq. mi., the quantity of rain in each hour is reduced by a given percentage to reflect the fact that the air mass containing these rainfall characteristics has a finite width.

divided by lake surface area. Use of this measure in conjunction with an examination of the geotechnical character of the sill permits a preliminary assessment of the sill's stability. If the sill could be disturbed by the calculated rise of the lake level, then the actual rise of lake level should be determined taking watershed infiltration and lake outflow into account.

Over the long term, the volumetric buffering capacity strongly influences downstream flow characteristics and their associated transport and erosional characteristics. These characteristics are important if radionuclides associated with particulates escape from the lake due to atmospheric extrema. If the downstream waters are depositional in character (e.g., a lake with a small watershed area but with a large surface area), it will be relatively less affected by the atmospheric extrema and may trap all eroded materials from the upstream lakes.

The long-term lake level influences the stability of the overflow sill, the actual discharge nature of the lake, and the potential exposure of people to the tailings. Long-term levels which are higher than present provide higher hydraulic forces (called the hydraulic head) for destabilizing a sill. If the long-term level is lower than present, the lake may only discharge intermittantly. Many lakes in southern Saskatchewan have this character (e.g., Old Wives Lake). They will discharge when a sufficiently large rainstorm (e.g., a 1-500 yr.) fills them up and causes them to flow over their natural discharge sill. At all other times, evaporation exceeds inflows (groundwater flow is significant for some lakes).

Lower lake levels would permit human access in the future to lake areas closer to tailings. For example, if redistribution of the tailings occurs due to lake currents, if the tailings are carried into nearshore zones, and if the lake level lowers, humans might be exposed to such zones. Long-term lake levels are strongly influenced by long-term climatic variations. An interesting example is the long-term variation of the level of Lake Geneva. For several years, anthropologists theorized about the sociology, the ecology and the food-gathering characteristics of humans who lived in houses on stilts in 10-20 feet of water in Lake Geneva. However, examination of lake sediments has shown that during this period in time, the lake levels were quite low because of a drier climate. These people were, in fact, living on the shore of the lake; the broken-off poles still in the lake sediments were part of the structure of huts which were preserved until today by being under water.

6.4.5 Lake Mixing Characteristics

Measurements

1. influence of inflow characteristics upon lake mixing
2. disturbance of lake sediments by extreme currents

The inflow characteristics of rivers can control the horizontal and vertical characteristics of lake depending upon their inflow rate and the time of year. They can influence the extent of erosion of sediments. Wind-driven mixing arising from high winds is often sufficient to disturb and to mix sediments.

Inflowing water mixes with lake water to some extent and then seeks its own density level. To illustrate the effects of such inflow on lake circulation, three examples are given.

Example 1

If the density of the inflowing water is higher than bottom waters, it could flow to the bottom layer and displace bottom waters.

Example 2

In the springtime in Canadian Shield Lakes, 1°C water from the rivers may flow under the ice surface across to the outflow. It acts as a river flowing through the lake and only mixes a small amount. If the majority of spring runoff occurs before ice-out is completed the effect of inflow on a lake's constituents is relatively small. If the majority of spring runoff occurs after ice-out and if wind driven mixing is substantial, the inflow waters will mix with all lake waters. If, however, there are insufficient winds for one to two weeks after ice-out, the surface water temperatures will heat rapidly such that surface waters are less dense than bottom 4°C water. This would reduce the effective mixing of inflowing waters with the full lake. Such runoff patterns have substantial implications for the vertical mixing patterns if meromixis is used as a control measure.

Example 3

If cold rain water enters a lake during summer stratification, its density may equal that of the thermocline. Inflows to the thermocline

tend to decrease the stability of stratification. If inflow is to the surface and withdrawal is from the bottom (e.g., a bottom-draw reservoir), vertical transport may be sufficiently high to prevent stratification.

A parameter useful for assessing the vertical stability of stratified lakes subject to large inflows is a lake Froude number.

$$Fr_L = \frac{LQ}{V} \frac{g\Delta\rho}{\rho_o} Z$$

This number relates the inertial forces due to throughflows to the gravitational forces due to stratification (L, z, V - length, mean depth, volume of reservoir, Q - rate of throughflow, Δp - density difference between epiliminion and hypolimnion, ρ_o - reference density and g - gravitational acceleration). If $Fr_L << 1/\pi$, the lake will be stratified. If $Fr_L >> 1/\pi$, the lake will be vertically mixed. Lakes which approach a vertically mixed character are typically well down in a chain of lakes and are shallow.

Large inflows will also aid in eroding shallow sediments. The history of sediments in nearshore areas will aid in assessing this point.

While large inflows associated with rainfall have substantial effects upon vertical lake stability, the lake stability is doubly threatened by the association of high winds with rainfall-runoff events. In fact, large winds by themselves are sufficient to destabilize a lake during summer stratification. The timing of the wind-runoff event is also important - intense summer stratification is usually able to resist large winds of short duration.

If wind energy or combined wind-runoff energy is strong enough, bottom waters may mix sufficiently to disturb bottom sediments. This occurs particularly during unstratified conditions. Examination of sediments for evidence of past mixing allows statements concerning the frequency of such mixing and the intensity of mixing (e.g., a vertical concentration profile in sediments over a depth of a few mm to cm for an element whose profile is normally sloped). Evidence of such mixing permits statements about the need for a tailings barrier or a sediment barrier above the tailings.

6.4.6 Transport from Lake to Watershed Components

Possible Measurements

1. exhalation of gaseous radionuclides from lake
2. possible routes for transport of radionuclides to land-based animals

Under conditions of deep lake disposal of tailings, it is generally assumed that aqueous diffusion of soluble nuclides, movement of soluble nuclides with currents, physical movement of tailings because of earthquakes, and transport in the biota will be the main mechanisms controlling nuclide transport. This contrasts to dry deposits of tailings in which atmospheric transport of particles and exhalation of gaseous nuclides such as Rn are important. If gaseous nuclides can escape from the lake water, identification of the quantities which may potentially escape, of atmospheric transport rates, and of receptors are in order as diagnostic tools for determining the importance of this potential transport route.

Potential transport of nuclides to terrestrial environments and food chains is an important consideration. If nuclides are transported out of sediments and are accumulated by algae, bioaccumulation up the food chain to fish is possible. If macroinvertebrates burrow into the tailings or contaminated sediments and ingest nuclides through feeding upon organics or bacteria, benthic predators (e.g., fish) may subsequently accumulate the nuclides.

The ecology of the biological community would determine the possible transport routes for bio-accumulated nuclides. The emergence of aquatic insects into terrestrial adult forms may represent one of the most important means of biological transport. Fish feeders such as birds or bear might be expected to be possible receptors. Fishing by people is another transport route. Swimming of fish out of the lake to inflowing streams or downstream waters are routes for nuclide transport to animal feeders in other environs.

Rooted plants such as macrophytes can absorb soluble constituents such as nutrients and radionuclides from water. If ingested by animals, these nuclides can be transported to other areas of the watershed.

These possible routes can be quantified using dose calculations such as those made by BEAK (1979, 1981). Such calculations permit one to define the extent of any potential problem by comparison of incremental ecosystem concentrations due to tailings disposal with concentrations attributable to background conditions.

6.4.7 Perspective on Atmospheric Interactions

Atmospheric interactions with a lake (heat effects, wind effects) mainly control the vertical and horizontal mixing characteristics of a lake. These effects are diminished by the protection provided by surrounding hills; they can be amplified by the effects of runoff upon lake physics. The strength and efficiency of energy transfer influences the energy of bottom currents and the percentage of bottom sediment areas which are erosional in nature.

Mathematical tools appear to be capable of realistically predicting vertical transport characteristics in lakes. These predictions can be made when stratification-caused vertical density differences and wind energy one metre above the lake surface are known. A major problem, but one which is rapidly being solved, is the effect of the atmospheric boundary layer upon wind velocities above the lake and the influence of variations in boundary layer's properties between the shore and open waters. Mathematical tools are also available for predicting horizontal currents in a realistic fashion.

Dimensionless numbers (e.g., Lake Froude Number) are available which describe gross stratification characteristics of lakes. They can be used to examine such questions as 1) will leakage around the side of thermocline occur? and 2) will inflow rates prevent the development of stratification? These numbers should be utilized before vertical transport of lake is evaluated using mathematical models of heat transport. Dimensionless numbers for determining whether a lake will remain stable as a function of wind energy are available, but not as a function of water clarity. This is particularly important if acidic lakes allow sunlight penetration to deeper depths than in nonacidic lakes.

Precipitation and its naturally acidic character drive the weathering reactions of watersheds. Atmospheric inputs of materials and watershed export (weathering reactions, modified by humans and terrestrial biota) generally control the actual chemistry of lake water and its biota. The material budgets for a lake and their probable changes provide a tool for determining the long-term chemistry of lake water and its biological regimes over the next 10 000 years.

The future ability of aquatic biota to remove soluble nuclides from the lake water by uptake and subsequent settling is then fundamentally controlled by future changes in material budgets and weathering reactions. The adequate estimation of future material budgets is, and will continue to be, a weak link for forecasting the suitability of lake systems as disposal sites for tailings.

Shifts in material budgets can probably be bounded by some creative "digging" and synthesis of the literature. An even more key area will be predicting the subsequent shift in aquatic biota and their ability to remove soluble contaminants from the water due to a change in material budget.

7.0 HUMAN INTERACTIONS

Future human effects on both the hydrological unit and upon a tailings-sediment system may be substantial, but their actual predictability becomes less tractable the longer the time unit of concern. In this section, it is assumed that long-range climatic variations will occur and that human activity may be one of the causes. However, continuing the philosophy (section 6.4) of differentiating what will be a local or regional effect (e.g., trans-drainage basin transport of pollutants) from a global effect is one method for differentiating the relative importance of human actions, and, hence, whether it is reasonable to include such human effects as a question relevant to this project. BEAK makes no final judgements on this point, i.e., we do not draw a line.

The definition of the location of this "line" will continue to evolve in the future. It will, in fact, evolve as a result of detailed analyses by future proponents of lake disposal of U-mine tailings. For example, analyses of a tailings-groundwater-sediment system in one lake even if perturbed by annual earthquakes may show that a minimal leaching rate will be attained. Alternatively, some lake systems may exist where acid precipitation changes the lake pH sufficiently that significant quantities of material leach out; this may be true particularly in a lake with a watershed having negligible future erosion, even if a one-metre barrier is placed around the tailings. A third possible scenario is that leaching may become significant because man-induced climate changes block out significant amounts of solar radiation causing algal populations to shift and reduce their sequestering ability for soluble radionuclides.

BEAK suggests that the detailed analysis of a proponent would draw the line in the first example to exclude most of the sub-systems considered in this project (including lake water). In the second example, human effects would need to be considered. However, this example does not specify whether or not natural background levels would increase due to watershed leaching of an exposed mineral lobe containing material similar to the tailings. Under an assumption that the long-term background is the appropriate criterion for scaling the net release rate (the NR/W ratio of section 2.2), the detailed analysis by a proponent may show that NR/W is acceptable. In the third example, it may be reasonable to assume that other problems of more significance to human populations than leaching from tailings may occur as a result of these changes. If so, then the "line" is drawn not to consider this

human-induced effect. It also may be drawn to exclude this possible effect because of the many unknowns associated with predicting variations in climate due to man's activities.

It is suggested that drawing of the "line" in the case of the third example may be made with the aid of theoretical "what if's" and scenario testing. Further, it would be helpful for Canadian Society as a whole (whether politically or possibly through the AECB as the policy decision body) to draw such line a priori in the case of the third example. This is similar to what was done for this project by limiting the "reasonable" time scale of concern to the Canadian Environment to 10 000 years. Whether the second example represents a case where a "line" should be drawn a priori is not clear. General scenario testing would be quite useful for deciding this point or at least deciding some general atmospheric characteristics which should be considered by lake disposal proponents. Perhaps this latter point is most appropriate as a suggested policy methodology.

Long-term prediction of future atmospheric characteristics will be a general problem while most of the remaining aspects of the lake disposal question are more easily boundable. A methodology might involve setting a range of general atmospheric characterisicts (1-4 possible combinations) which should be considered by the proponent. However, until such characteristics are derived, it is suggested that a proponent would need to consider alternatives; perhaps use, on their part, of a "brainstorming" session of various experts may provide them with suitable hypotheses for selecting future atmospheric characteristics for analysis. Of course, as example 1 suggests, analyses incorporating atmospheric-drainage basin effects may not be needed for all lakes!

7.1.1 Human Effects upon Atmosphere

Major forseeable human effects are climatic change (wind patterns, rainfall rates) and chemical changes. Only the latter are considered here. Whether the former should be considered in lake disposal is probably a question of balance (relative importance of questions) as suggested above.

Further, it is suggested that chemical changes such as CO_2, particulates in the atmosphere, etc., which have a global significance may belong in the class of questions whose balance needs to be considered. Accordingly, they are not considered here.

The major atmospheric changes, caused by man, which may need to be considered by a lake-disposal proponent are changes in atmospheric chemistry which control long-term weathering reactions. Elements of concern include CO_2, SO_2, NO_3^-, and P. Radionuclides cannot not be disregarded if significant regional transport is foreseeable.

The first three elements are chosen as examples of the ability of precipitation to attack bedrock or soils (i.e., affects on weathering reactions) and to affect the drainage basin terrestrial ecosystem. Under natural conditions, CO_2 will control the acidity of rainfall; rainfall will be buffered by any particulates in the water droplets. The addition of SO_2 and NO_3 from the combustion of fossil fuels and from other sources now control the acidity of rainfall in many areas of Canada; the extent of the problem has only been recently realized. Since this acidity is dominated by man's inputs into the atmosphere, the extent of future acidity needs to be scoped for analysis of the lake system. A possible method (probable fuel consumption and its rate in the future) is suggested in section 6.4.

The element phosphorus is suggested because it is a controlling nutrient for biomass formation in many lakes. As atmospheric inputs of P are of equal importance to drainage basin inputs in many headwater lakes, future atmospheric changes (e.g., due to changes in agriculture, regional forestry practices) will strongly affect the development of biomass and hence the lake's ability to form particles which sequester mobile radionuclides. There is one qualifier on this example - it assumes that atmospheric P is biologically available for algal growth. Some work has been done on this availabiity question which suggests that it is a reasonable assumption; however, more work is needed. In addition, the future impact of NO_3^- changes upon lake biota where NO_3^-

is acting as a nutrient cannot be ignored. In lakes which are nitrogen-limited, higher NO_3^- inputs from the atmosphere may change it back to being phosphorus-limited. Associated shifts in algal populations may change the general sequestering ability of the lake's biota. The effects of P upon the drainage basin ecosystem may also be important; examination of drainage basin elemental budgets and fluxes (e.g., Likens et al, 1977) would be useful for analysis of this example.

7.1.2 Human Effects upon Drainage Basin

Major foreseeable effects upon the drainage basin include forest fires, logging and agricultural or urban activities. Forest fires could denude the watershed. Logging will remove large quantities of vegetation. Both effects should be temporary (e.g., 10-100 yr.) depending upon the rate of natural rate of revegetation and human efforts to reforest the drainage basin. Data such as that of Likens et al (1977) permit statements concerning the actual change in watershed characteristics and watershed export. While it is plausible to argue that such changes would be significant only during a first period (e.g., first 100 yr.), they could be important 500 years hence if the natural geochemical sequestering mechanisms in the tailings break down and allow substantial materials to migrate through the lake sediments into the lake water. The relative timing of such occurrences and the importance of watershed export upon lake sequestering properties are the most relevant factors in this question.

Agricultural activities cause significant changes in watershed export. Such changes could have positive or negative effects.

Urban activities could involve similar denudation effects on a fairly large scale or on a small scale. Small scale activities include driving across the watershed with recreational vehicles and the use of recreational camps in the watershed.

On balance, control of the majority of these effects depends upon planning controls and their long-range viability. Planning controls mean the zoning of land to confine its use to certain general kinds of activities. They have only recently been used in Canadian history,

beginning in Common Law and in religious canons and being recently formulated in the form of legislation. Their continuance generally is dependent upon political will for enforcement. How long planning controls could continue (10 000 yr.?) is dependent upon the stability of the society and evolution of future concepts of land-use control. Accordingly, a fairly philosophical question may be raised. For what time period is it reasonable to assume that present concepts of planning controls will be maintained and for what time period is it reasonable to assume that future political institutions would be able to maintain these controls? Some partial answers (100 yr.) can be ascertained from the general directions of population development in Canada - for example, the majority of population is located near the American border and near bodies of water while interior towns develop based upon resource extraction (minerals, energy, agriculture). If resource extraction ceases, will the human population move on?

The necessity for planning controls will depend upon the time frame and assumptions of watershed behaviour necessary to justify the viability of disposal in any one lake (e.g., if the "line" justifying disposal in a lake is dependent upon maintenance of watershed inputs of certain types for 1 000 yr., will future human effects on the watershed change these inputs in a beneficial or detrimental manner?)

If planning controls are judged to be crucial but too difficult to maintain near future areas of human encroachment, another model for management relates to the Experimental Lakes Area where radioisotopes were inserted in a whole lake experiments in an isolated area. This model suggests that a tractable approach for tailings disposal would be to select a lake in an area of Canada which is sufficiently isolated that a 200-500 mi. buffer zone may be created within which extraction of resources is allowed but no human habitation is permitted.

7.1.3 Human Effects on Lake

Foreseeable examples for this section are alteration of the outlet level of the lake (construction of a dam, dynamiting the overflow sill), boating activities leaving oil spills or creating large bottom currents (e.g., shipping), addition of chemicals to maintain pH, and insertion of poisons which kill or alter the lake biota. Such effects are probably insubstantial with two exceptions. Addition of poisons (whether accidental, or planned) could cause a significant change in

the sequestering behaviour of the biological particulates. A large lowering of lake level sufficient to dry it up could expose the former sediments to the atmosphere. Present day planning for lake selection should avoid lakes which might be used by large lake freighters.

7.1.4 Tailings-barrier-sediment system

Direct effects upon the tailings-barrier-sediment system by humans would relate to such physical effects as vandalism or chemical affects such as direct injection of a chemical into the sediment system. Indirect effects would be caused by direct human effects upon the atmosphere, the drainage basin or lake water which are transferred through the lake as described earlier in this report.

The probability of occurrence of such physical perturbations as vandalism is difficult to assess, particularly since the physical agent which might be used (diver with a small shovel, man in a boat with a weight attached to a fishing rod, dynamite released from lake surface) is quite unpredictable. Use of management tools such as ensuring minimum depths of water over the tailings system (e.g., 10 m, 20 m, or 30 m, e.g., depths of water to which a snorkler cannot reach, or depths of water to which a diver cannot return without decompressing) would provide some "buffer" to minimize such damage. For lake design, a reasonable vandalism scenario (e.g., a depth to which an average snorkler cannot reach) might be chosen by the board as a flexible constraint for evaluating proposals of a proponent.

Assessment of these detrimental possibilities is difficult; it may be one of the more significant "fears" which arise in the future in the area of public concerns. Scenario testing (hypothesizing "what if's" and quantifying the results) may be one approach to minimize such fears; whether it is reasonable for a proponent to consider or for a generalized study to consider is unclear. Analyses of the future binding ability of a sediment subject to earthquakes would give information similar to that needed to assess physical disturbances by vandalism; analyses of the binding ability of the sediment subject to acid lake conditions would provide information needed for "chemical" vandalism involving similar materials.

7.2 MITIGATING MEASURES

Mitgating measures may be viewed as methods for minimizing the potential for tailings leachate migration in certain directions. A main concern that arises from insertion of tailings in a lake system is the migration of radioisotopes into the food chain and the associated biological damage. The two main pathways are migration into groundwater (will it subsequently come to the surface or not?) and into surface waters. A tailings barrier may be viewed as an attempt to minimize transport into surface waters and thus direct transport into deep groundwater areas. Particularly where groundwater gradients (below 4 m) are vertical (e.g., areas of Northern Saskatchewan), placement of a barrier above the tailings but not below the tailings and suitable deep bedrock permeabilities (somewhat higher than intermediate strata) would force dispersal into the deep bedrock areas. For isotopes with half-lives of the order of 80 000 yr. (e.g., Th) and for time frames of the order of 5-10 000 years, substantial quantities will diffuse (solid-state diffusion) through 1 mm particles into pore water and be transported somewhere; the direction is a function of the direction of groundwater flow. Insertion of a barrier whose permeabiltiy is lower than that of bedrock would aid in directing flow into deep bedrock. Then one is using the assimilative capacity of the bedrock to minimize transport into surface waters.

Mitigating measures are those which either induce changes in the natural long-term characterisitcs of the watershed lake system or engineering designs which are capable of increasing the containment potential along one particular transport pathway (e.g., barriers, engineered chemoclines). The usefulness of each mitigating measure may be assessed by using the following "Mitigating Measures Methodology":

1. The biochemical or geochemicals system(s) and the transport pathway(s) influenced by the mitigating measure should be determined.

2. The containment potential along one pathway and hence the dispersal potential along other pathways should be determined. For this purpose, the organization used in this report (hydrological units, transport routes) is useful.

3. The assimilative capacity of the receptor environments could be determined.

4. The biological significance of levels of contaminants in receptor environments should be determined.

This report now examines several alternative mitigating measures and makes some statements about their short-term versus long-term effects. The analysis proceeds using the methodology outlined above. Only brief statements will be made about points 2 and 3 as they will be site-specific and design-specific. The term "transport routes" of point 2 means following the routes through which material can flow (for example from the tailings to the downstream receiving waters). Varying degrees of emphasis are placed upon different mitigating measures. This is not intended to mean that any one measure is a potentially better measure but rather that we chose, in the limited time available, to examine some mitigating methods which appear to have potential but which have been relatively less well-evaluated.

Firstly, "Engineering of the Lake" measures are described. Then, "Environmental Engineering" measures are described next. Then, meromixis is described in detail as it is potentially both a pit engineering and an environmental engineering measure.

7.2.1 Engineering of the Lake Mitigating Measures

Possible Methods

1. addition of cementing agents to tailings
2. method of tailings placement
3. creation of barrier around tailings
4. creation of meromixis
5. bypassing inflowing streams to outflow
6. creation of artificial hills around lake
7. planting trees nearby

Addition of cementing agents to the tailings could involve addition of chemicals which cause additional chemical precipitation or which increase pore water concentration of ions needed to maintain existing chemical precipitates and which would cause these precipitates to form relatively hard-to-solubilize crystalline forms. Such chemicals include sulfate required to maintain a $RaBaSO_4$ precipitate. Knowledge of the desorption mechanism in the tailings would aid in this "geochemcial engineering" mitigating measure.

The methods of tailings placement is important if different placement methods cause different initial permeabilities or other physical-

chemical properties in the tailings or lakewater which would be maintained over the long-term. Alternative placement methods which might be considered include barge dumping, hypolimnion injection, and removal of water from the lake during filling. Barge dumping is effective, if the tailings stay intact during the time of settling from the lake surface to the sediments; if not, significant leachate may remain in the water column. In this case, the long-term question is whether other measures such as artificially fertilizing the lake create sufficient sequestering agents in the form of biological particulates to remove residual leachates over the short-term. Use of information from dam-building (construction of earth-fill dams using barges on a reservoir behind a coffer dam) and information on leaching from barge dumping of solid waste in near-shore oceanic areas (e.g., New York city, Los Angeles, etc.) would be of some aid here.

Hypolimnetic injection would direct the tailings to its final resposi-tory and minimize the exposure of surface waters to pot leaching. The placement of tailings would require movement of the pipeline to assure even distribution; such movement is often not feasible, resulting in mounding and erosion-deposition phenomena. Alternatively, barging allows better location of the placement of the tailings. Dry lake filling allows visual inspection of the quality of placement of tailings; the expense of maintaining dry conditions may be prohibitive for many lakes compared to the additional benefits.

Placement of a barrier such as clays minimizes pore-water migration of leachates to the lake sediments; in addition, typical clays have a high adsorption capacity for many radionuclides and act as an effective retardant. Certain problems arise from underwater placement of clay barriers. A uniform clay has particles which are able to repulse (electrical charge-wise) each other (because of the "double-layer effect"). Such-repulsive charges cause bridges to form between particles, preventing adequate consolidation and leaving large void spaces. Heterogeneous clay mixtures of different electrical charge properties aid consolidation. Furthermore, particles like apatite (calcium fluoro-hydroxy phosphate) have shown an even higher adsorptivity for radionuclides than clays; in the case of apatite (and possibly even for clays), permanent coprecipitation of a radionuclide may occur. Liberal inclusion of apatite into a clay liner expands the retarding effect of a barrier in migrating radionuclides.

The question arises as to whether a barrier should be made of concrete (containment) or of clay. For the time period that concrete is stable, it is a very effective containment device. However, an earthquake or other geotechnical movement or aggressive groundwater can cause the concrete to break up. If a rupture occurs, then a potential for a high

bleed rate has developed. On the other hand, geotechnical movement may cause the clays to shift and allow some contaminants to escape. However, they should subsequently reconsolidated, minimizing further escape of contaminants. Hence, the question is one of use of a rigid containment system as opposed to an elastic system. The elastic system probably allows a higher bleed rate, but the risk of total barrier failure is much smaller compared to the risk associated witha rigid system.

Meromixis or chemostratification involves the addition of soluble salts to bottom waters to cause a permanent density stratification. Conceptually, it offers the ability to prevent wind-driven mixing from reaching bottom waters and hence, it would prevent erosion of sediments and vertical transport in the water column due to advective type processes. The only vertical transport process of substance in the water column then would be diffusion. Further, meromictic waters are anaerobic. This favours the precipitation of metal sulfide which may be capable of coprecipitating migrating radionuclides. Sequestering and settling of radionuclides by biological and inorganic particles are additional safety mechanisms, if nuclides diffuse to the upper layers of the salty waters. Engineering of a pit involves selection of geometry to minimize vertical circulation and placement of salt in the bottom waters of the pit.

As meromixis is a strong option as a mitigating measure, a detailed evaluation of lake physics (section 2.1) and of atmospheric effects upon the lake (6.3) have been given above to describe forces causing vertical mixing. Observations on natural mechanisms for the formation of meromixis and upon artificially created meromixis are given below.

Bypassing streams around a lake would be quite useful for short-range and long-range problems, but has some drawbacks. Bypassing permits the lake level to be controlled by the precipitation-evaporation balance if groundwater flows are insignificant. In many non-marine areas of Canada, evaporation exceeds precipitation and no net outflow would occur, unless significant land-based or groundwater inflows exist. Typical areas include the prairies, lee areas of the Rockies and northern Canada. In areas like southern Ontario, precipitation approximately equals evaporation due to the influence of the Great Lakes. In seacoast areas of the Rocky Mountains and Newfoundland, precipitation greatly exceeds evaporation.

During the short term, bypassing streams prevents discharge of tailings leachates to receiving waters if such leachates build up in the lake water during the time period of tailings placement. Over the long-term, bypassing streams would likewise prevent entry of any tailings

into downstream surface waters and aid in preventing river associated plumes from destabilizing meromictic water. The drawback to such a bypass is that it decreases the nutrient supply to a lake and hence decreases the biomass available for sequestering any soluble radionuclides. Inputs of nutrients from the atmosphere would be the main source.

Creation of artificial hills decreases the effects of winds and hence the amount of vertical lake transport. Some aspects of this point are discussed in Section 6.3

Planting trees decrease the effects of winds, similar to but in a less efficient manner than the effects of hills.

7.2.2 Environmental Engineering Mitigating Measures

Possible Methods

1. change land use patterns of drainage basin
2. add nutrients to a lake
3. neutralize watershed or lake
4. meromixis

Environmental engineering measures (or geochemial engineering measures) for mitigating the effects of lake disposal of tailings are defined here as those in which the ecosystem is artificially manipulated to cause a change in the biogeochemical regimes of the watershed-lake system. Such measures may be short-term (e.g., aeration of a lake to cause turnover and prevent the formation of anoxic waters) or long-term (e.g., reforesting a watershed to decrease nutrient inputs into a lake).

A change in landuse patterns can induce two types of changes in a lake, depending upon the objective. Reforestation will decrease the input of particles and nutrients while periodic logging or agriculture will increase the input of particles and nutrients. If the objective is to create more organic and inorganic sequestering agents, watershed disturbances are desired. If the objective is to prevent the development of hypolimnetic anoxia, then reforestation will reduce nutrient inputs, causing a reduction of biomass which settles to hypolimnion waters and demand oxygen due to bacterial oxygen consumption. Further reforestation may reduce the dominance of algal-type sequestering organics and replace them with humus-type sequestering organics.

Temporary or permanent addition of nutrients to a lake (e.g., fertilizer or sewage) will aid the development of algal-type sequestering

organics. Excessive additions may cause the hypolimnion to become anoxic during stratification.

Due to acidic precipitation, the lake water may become increasingly acidic over the long term. This may result in a shift in aquatic plants from floating algae to algal mats on the lake bottom (acid resistant, but able to obtain solar radiation due to increased water clarity) decreasing the sequestering ability of algae. Acidic waters may penetrate to the tailings (if no barrier is present) to increase leaching of contaminants. Neutralization of the lake with lime has been attempted in some Ontario, Swedish and Polish lakes with some success; but the pH always decreases due to continued acidic precipitation. Neutralization of watershed streams (insertion of limestone in water courses) is necessary to provide some long-term benefits. The length of time over which this would be effective will be evaluated in the near future by several agencies (e.g., Scandinavian, Canadian, American).

Meromixis itself is an environmental engineering method as it may substantially change the lake ecosystem. Due to the decreased transport of oxygen, the chemostratified layers could become anoxic due to decomposition of settling organics. As described in section 2.1.5, this aerobic decomposition could release adsorbed materials from settling organics. Anaerobic decomposition is normally slower. Further, anoxia may act to mobilize or to sequester further radionuclides migrating through the sediments. If the adsorption capacity of clays decreases under anoxic conditions, this acts as a mobilization mechanism. If $RaBaSO_4$ or tailings are leachable under anoxic conditions, radionuclides may be mobilized. However, if iron sulfides are formed at the sediment-water interface and coprecipitate radionuclides, then axonia aids sequestering mechanisms in the lake. Examination of section 4.0 provides some information on this point; natural and artificial controls on meromixis are discussed next.

7.2.3 Natural and Environmental Controls or Meromixis

7.2.3.1 Natural Control of Meromixis

During time of transition from stratification (e.g., summer) to stratification (e.g., inverse winter), periods of vertical circulation or mixis occur. The causes of this circulation are evaluated in section 2.1, "lake physics". Descriptive terminology is reviewed here. The great majority of the world's temperate lakes undergo mixis twice annually, exhibiting an autumnal circulation as colder air temperatures destroy the summer stratificaiton and a vernal circulation as warming breaks down the winter thermal structure. In both cases, the density

difference between surface and bottom waters decreases to a point where surface wind energy is sufficient to mix the water column entirely, bringing bottom water to the surface and surface water to the depths, hence, the common term, turnover, for these periods of mixis.

Lakes which undergo both a spring and a fall turnover are said to be dimictic, many Canadian lakes fall into this class. In tropical or arctic latitudes, other thermal lake types occur such as monomictic (one turnover per year) and polymictic (several turnovers per year). Monomictic lakes may be further subdivided into cold and warm monomictic lakes. Cold monomictic lakes (e.g., in the Arctic) have only one period of circulation in the summer at or below 4°C, warm monomictic lakes (e.g, in warm temperature-constant regions of N.A. and Europe) circulate freely in the winter at or above 4°C. The term monomictic is occasionally applied to Canadian lakes where temperatures rise quickly in spring and winds are calm so that surface heating and thermal stratification occurs without any initial period of turnover. All these types are similar in that mixis regularly occurs and entails a complete turnover from top to bottom; such lakes are classed as holomictic lakes.

Findenegg (1935) identified a smaller group of lakes in which some water remains partly or wholly unmixed with the surface water mass. These lakes are called meromictic; the deeper, unmixed layer is called the monimolimnion. Hutchinson (1937) introduced the term mixolimnion to describe the well-mixed surface waters and the term chemocline to denote the zone of sharp density gradient separating the two layers. Meromictic lakes are relatively rare, but they do occur in many regions of the world. Canadian meromictic lakes were first discovered in coastal regions (Williams et al, 1961); Northcote and Halsey (1969) were the first investigators to report meromixis in inland Canadian lakes. To date, 27 such lakes have been described in Canada: 11 in Ontario, 7 in British Columbia, 4 in the Northwest Territories, 2 each in Saskatchewan and Quebec, and one in Labrador (see Table 7.2.1). These numbers will undoubtedly increase as more lakes are investigated.

Meromictic Lake Typology

Since meromixis is usually caused by density gradients due to factors other than temperature (e.g., salinity) the tendency has been to class meromictic lakes according to the presumed causes of their salinity gradients (picnoclines). Findenegg (1937) proposed two categories: static, if the picnocline is dependent upon the geologically determined presence of saline water (usually involving sodium, chloride and sulfate ions); or dynamic, if biochemical decomposition and dissolution

of the sediments leads to increased concentrations of solutes such as bicarbonate and silicate in lower waters.

Hutchinson (1937) divided Findenegg's (1937) static category into two classes and suggested a threefold classification:

1. Crenogenic meromixis (static) is due to saline springs bringing dense water to the lower layers of a lake. If the saline spring supply is considered to be continuous over a long time scale, then this type of meromixis can be seen as permanently stable. Crenogenic meromixis is usually associated with salt-bearing geologic deposits.

2. Ectogenic meromixis (static) is caused by an external catastrophe bringing salt water to the bottom of a fresh water lake or an external input bringing fresh water to the top of a salt water lake (e.g., The Dead Sea). This usually occurs in coastal lakes, involving storm-driven incursions of sea water into lake basins. Temporary ectogenic meromixis is found where periodic spring inflows bring in relatively salty water, compared to the normal pattern. Following the second law of thermodynamics, this type of meromixis may ultimately degenerate if wind energy is sufficient to eventually erode the stratification, unless the catastrophic events occur frequently enough or another source of stability is also present.

3. Biogenic meromixis (dynamic) is due to accumulation in the monimolimnion of salts liberated from the sediments as previously described. Once the monimolimnion has been established, biogenic meromixis tends to increase in stability with time.

   Working form a much larger base of information, Walker and Likens (1974) expanded the classification to include five types:

   i) Ectogenic, as defined by Hutchinson (1937).
   ii) Triptogenic meromixis. This is due to surface inflow of turbidity currents, such as may occur after the clear-cutting of a forested watershed.
   iii) Biogenic (Hutchinson, 1937) or morphogenic meromixis Northcote & Halsey, 1969). This originates by virtue of wind shelter afforded by the shape and size of the lake basin and by local topography, and in most cases is aided by chemical stratification contributed by biological processes. The relative importance or morphometry and biogenesis depends

on the characteristics of each individual lake. Many of the world's very deep lakes are practically morphogenially meromictic (i.e., they only infrequently circulate to their depths).

iv) Crenogenic, as defined by Hutchinson (1937).

v) Cryogenic meromixis. This is due to deep-water accumulation of salt precipitated by "freezing out" from a surface ice layer. Cryogenic meromixis has been observed in Antarctica (Goldman et al, 1972) and may well exist in the high Canadian Arctic.

It should be noted that meromixis is often due to a combination of two or more of the five above mentioned types.

Calculating the Stability of Meromixis

Hutchinson (1937) defined meromictic stability as the amount of work needed to mix a meromictic lake to a uniform concentration if it were closed, or to mix the monimolimnion into the perennially freshened mixolimnion if open, i.e., the amount of energy needed to make the lake holomictic. The stability lies in the difference between this amount of energy and the total energy actually available, primarily from wind exposure and solar radiation but also possibly from turbulence caused by inflowing streams and conceivably, in very small lakes, from motor boats. The amount of work needed can be calculated from the density of the saline layer and the morphometry of the basin, and is usually expressed in units of g.cm cm^{-2}.

Hutchinson (1937) used this method to explain the degeneration, over a 62-year period, of meromixis in the Hemmelsdorfersee. The lake was originally made meromictic by a catastrophic incursion of sea water in 1872, but gradual mixing between the monimolimnion and the mixolimnion caused the chemocline to descend gradually until it had disappeared by 1934. The original stability, in 1872, was calculated to be 2 690 g-cm cm^{-2}, an enormous amount of energy, but when spread out over 62 years, a quite reasonable mean value of 44 g-cm cm^{-2} was required. By this type of calculation, the salinity required to produce meromixis could be calculated if the amount of available energy, or a probability distribution of it, were known.

Indications of Meromictic Conditions

The only definitive means of establishing that a lake is meromictic is to regularly measure its temperature and conductivity profiles over at least one annual cycle to ensure that it does not mix. The effort required for this method would preclude its use in many situations especially those involving exploration of remote regions. There are, however, many conditions easily identified in one visit that indicate a strong probability that meromixis exists:

1. Existence of a picnocline. The presence of a strong vertical density gradient, often accompanied by a sharp chemocline, can be verified by a simple vertical conductivity profile.

2. Anomalous temperature curves. Holomictic lakes exhibit, during their stratified periods, a monotonic change in temperature with depth, either decreasing downward from the surface (summer) or increasing downward (winter). However, when sufficient vertical variations in the chemical composition of the water occur, relative maxima or minima may be observed within a stratified water column. When a minimum occurs the profile is said to be dicothermic, and when a maximum is present the condition is known as mesothermy. In very rare cases, one or more maxima may coexist with one or more minima, a state called poikilothermy. Dicothermy is particularly characteristic of summer profiles in meromictic lakes, where the temperature decreases with depth to a minimum at the chemocline, below which the monimolimnetic temperatures rise due to adiabatic heating from the sediments. Thus, the lower water is warmer, but yet heavier because of its salinity, so that it stays in the monimolimnion. Dicothermy can be easily detected with a submersible thermister.

3. Sulphur bacteria. The presence of sulfate in the monimolimnion often leads to anoxic conditions and to large concentrations of H_2S in deeper waters and a layer of purple or reddish-appearing sulphur-reducing bacteria in the chemocline. This is often readily visible, as in Pink Lake, Quebec (Dickman et al, 1981).

4. Basin and watershed morphometry. No natural meromixis can exist unless a lake is morphometrically predisposed to resist wind-driven mixing. The degree to which a lake basin is protected from the wind by its watershed is often intuitively evident, but more difficult to quantify. Relationships such as those of Hansen (1979) are one method for estimating the effects of sheltering. Since steep-sided watersheds tend to have areas that are

small relative to the surface area of the lake itself, the ratio of watershed area to lake area has been suggested (Olszewski, 1951) as an index of sheltering. Such an index would be useful only for drainage basins of severe terrain (e.g., mountains, hilly areas).

The depth of a lake in relation to its surface area is also important in resisting mixis. Walker and Likens (1975) have proposed the relative depth (Z_r, calculated by expressing the maximum depth, Z_m, as a percentage of the mean diameter) as a means of quantifying depth in relation to lake area. Lakes that are relatively deep should thus be more disposed to become meromictic.

Table 7.2.1 summarizes all known Canadian meromictic lakes are summarized for which suitable data are available. The ranges of latitude (43°-75°N) and longitude (62°-125°W) encompass most of Canada. Some lakes (e.g., Powell) are coastal, while others are well inland (eg., Deadmoose). The range in area (1.75-2 760 ha) and in depth (358 m, Z_m) is also great, but it should be noted that no truly shallow lakes are known to be meromictic. The most consistent parameter is Z_r, the relative depth. Even the very small lakes are rather deep for their area. For the 17 lakes for which we have data, 16 have a Z_r greater than 1.0. The only exception, Waldsea, is so strongly saline (crenogenic) that morphometry is probably not as important as in ectogenically or biogenically meromictic lakes, which are the two categories that include most of the lakes in the table.

7.2.3.2 Engineering Creation of Meromixis

Most Canadian lakes are not known to exhibit meromixis. However, many would appear to have the types of geometry required to minimize vertical mixing. Since many lakes are in the Canadian shield, it is presumed that the lack of meromixis is due to the lack of input of salts in substantial quantities into lakes whose geometry would appear to be suitable. Given the review of Atmospheric-lake interactions above (Section 6.3) the sheltering effect of nearby hills may be more influential than, or of equal influence to, lake geometry for determining meromictic conditions.

The approach of Hutchinson for calculating stability and rate of breakdown is one approach which may be used; it provides the basis for calculating but does not provide an ability to predict the rate of

Table 7.2.1: Geographical and morphometric data for some Canadian meromictic lakes. Reference numbers refer to literature sources given below.

Lake	Prov.	Lat. (N)	Long. (W)	Depth (m) Z_m	Depth (m) $\bar{Z}$	Area (ha)	Z_r* (%)	Ref.
Sunfish	Ont.	43°28'	80°39'	20	10.4	8.3	6.15	(1)
Crawford	Ont.	44°28'	79°57'	25	9.7	5.0	9.91	(2)
Pink	Qué.	45°51'	76°05'	20	12.4	11.5	5.23	(2)
Cayamant	Qué.	46°11'	76°19'	25		20.0	4.95	(2)
Nitinat	B.C.	48°42'	124°50'	205	99.6	2760	3.46	(3)
Yellow	B.C.	49°10'	119°45'	40	20.4	32.3	6.24	(4)
Mahoney	B.C.	49°17'	119°32'	19	7.2	21.6	3.60	(4)
Sakinaw	B.C.	49°38'	124°02'	140	43	800	4.39	(5)
ELA 120	Ont.	49°39'	93°50'	19	7.6	9.3	5.52	(6)
ELA 241	Ont.	49°40'	93°33'	12	5.5	1.75	8.37	(6)
Powell	B.C.	49°50'	124°32'	358				(7)
Lyons	B.C.	50°57'	120°07'	12		35.7	1.78	(4)
White	B.C.	51°22'	121°53'	15	4.6	99.6	1.33	(4)
Waldsea	Sask.	52°17'	105°12'	14	8.1	464	0.59	(8)
Deadmoose	Sask.	52°19'	105°10'	48	6.0	998	1.35	(8)
Tessiarsuk	Lab.	56°30'	61°57'	50		555	1.88	(9)
Ogac	NWT	62°52'	67°21'	60	23.2	148	4.41	(10)
Garrow	NWT	75°18'	96°50'	50	24.5	418	2.17	(2)

* Z_r is calculated as $50\ Z_m \sqrt{\pi} / \sqrt{A}$ %

1. Duthie and Carter (1970).
2. Dickman et al (1981).
3. Northcote et al (1964).
4. Northcote and Halsey (1969).
5. Northcote and Johnson (1969).
6. Schindler and Holmgren (1971).
7. Williams et al (1961).
8. Hammer et al (1978).
9. Carter (1961).

breakdown as a function of prevailing meteorological conditions. The wind conditions of the atmospheric boundary layer and secondarily, the heat budget must be included. For design, the following approach is suggested in a generic sense for evaluating the studies of a proponent who is considering the long-term stability of meromixes.

1. A general examination of area lakes could be made for the most suitable geometry (large depth, small surface area, pronounced relief in drainage basin).

2. Calculations could be presented (e.g., a preliminary design) showing probable salt concentrations needed. Within limits, density is dependent upon the mass of salt, not upon specific ions present.

3. Generic studies could be made (varying geometry, atmospheric protection effects due to watershed relief, salt concentrations) to develop nomographs relating the rate of breakdown of stratification to geometry (surface area, mean depth, atmospheric protection). As noted above, the transfer of wind energy is generally considered to be fetch independent and surface area independent, if the wind speed is known a meter above the lake surface area; the surface area parameter and ratio of drainage area to lake surface area, then, are measures more of atmospheric properties than of lake properties. Still, these parameters are useful surrogates as design tools.

4. For a few candidate lakes, the rate of breakdown of meromixis could be calculated, using their specific atmospheric boundary layer properties, their specific lake geometry and hydraulic inflow-outflow effects, if the lake were initially made meromictic. Selection of 20 years of atmospheric data (available from most master weather stations) plus estimation of the lake's hydrology based upon rainfall-runoff models would allow estimation of the long-term behaviour. The atmospheric properties could also be perturbed with the appropriate extrema (1-100 yr. wind, etc.) to evaluate their effects.

Tools useful for making these calculation are available; their application to existing meromictic lakes would increase the Board's confidence in their tractability and quantitatively explain meromixis in a better fashion than present. BEAK has used such tools in the past; satisfactory agreement between rate of entrainment and wind conditions has been obtained for thermal regimes but they have not been verified on meromictic lakes. The biggest difficulty associated with

such tools appears to be treatment of the atmospheric boundary layer; as described in section 6.3, this problem now appears to be tractable. It is presumed that calibration-verification studies, initiated both by government authorities and a proponent, would catalyze development of confidence in the utility of these tools. As some long-term studies (2-4 years) would be needed, the question of whether a project-specific orientated proponent could undertake these very long-term studies is posed.

7.2.3.3 Perspective on Meromixis

Meromictic lakes probably occur in most regions of Canada. More will be found as more limnological investigations are performed. Once initiated, ectogenic or triptogenic meromixis may be maintained by a combination of suitable morphometry and adequate levels of biogenesis of solutes in the sediments. Where they do not exist, the fundamental question of why must be faced - is it due to lack of suitable atmospheric effects, lake geometry, or salt source? Investigators searching for naturally meromictic lakes or for lakes with a good potential for artificially-induced meromixis, should look for lakes of large relative depth in well-sheltered watersheds. They should also be alert for the existence of picnoclines, dicothermy, and sulphur bacteria layers. The calculation of additional salinity required to initiate meromixis is relatively simple.

7.3 RELATIONSHIP TO BIOGEOCHEMICAL CYCLE

Human activity is, at present, one of the major catalysts for increasing the rate of and for modifying the direction of major biogeochemical pathways on a global scale. It is reasonable to expect this trend to continue in the future. The nature, the degree and the inevitability of human disturbances require assessment in order to examine long-term biogeochemical cycles, pathways and fluxes, and in order to estimate a geochemical baseline for assessing future tailings behaviour.

8.0 DEVELOPMENT OF CRITERIA FOR DEEP LAKE DISPOSAL OF URANIUM MILL TAILINGS

8.1 PROBLEM SETTING

As one method for management of uranium-mill tailings, a deep-lake disposal option is under consideration by various mining companies and regulatory authorities. Traditional methods have typically considered disposal on a surface site. Such a site suffers from several potential problems over the long term. Wind driven erosion may remove surface soils, allowing long-range transport of tailing particles, and exposing colonizing biological systems to radiation and other contaminants. Wind and water may erode these materials into downstream surface waters and deltaic soils. Extrema such as storms and earthquakes may make these surface sites more prone to erosion. Alternating wet-dry cycles may promote leaching from the tailings mass due to alternating states of redox (oxidized-reduced) conditions.

An appealing aspect of deep-lake disposal is that such deposition occurs into a net sedimentary, accumulating environment as opposed to a net erosive environment over geological time scales (e.g., 10 000 years). Overall, lakes tend to be environments of net deposition or "natural settling basins". Hence, disposal of tailings into a lake places such material into a geological formation which naturally does not erode in a net sense and which can withstand periodic extrema such as hurricanes and earthquakes without experiencing major perturbations.

One perceivable objection is that because most lakes mix from top to bottom annually, any material which does migrate from the tailings mass to the overlying water will be transported upward in the water column where the potential for affecting biological systems, including man, exists. The importance of this depends upon the rate of leaching, the direction of transport, and the effect that any periodic or long-term upset of the system will have on these intrinsic leaching-transport rates.

When compared with disposal on a surface site, lake disposal would be the preferred method for long-term management if more contaminant transport is directed away from surface waters and if the leaching rates to a lake are less. An additional advantage of lake disposal is

the net depositional character of a lake site compared to the net erosive character (over a 10 000 year time frame) of a surface site. The deep lake disposal option, through the placement of tailings into an environment that is physically, chemically and biologically more stable than a surface disposal system, ensures a more uniform and predictable loss of contaminants with time. The lake sedimentary environment also offers the potential of contaminant loss to deep groundwater flow systems and the consequent potential for very long-term sequestering (uptake) of contaminants by bedrock.

In this document, we have attempted to delineate the major controls on the long-term behaviour of both the source term (tailings inserted into a lake sediment) and the response of the open environment (e.g., long-term controls on pH, Eh, and "background concentration" of radionuclides and other tailings leachates) to a disposal system involving the placement of tailings into a sedimentary lake basin. From this delineation, a list of major and minor variables evolve which may be used for management analysis and regulatory criteria setting.

In this section, all variables considered in this report are tabulated and an abbreviated list of major variables is examined. These major variables should be considered to be the minimum set of variables which any proponent may need to consider in examining a deep-lake disposal option.

Several of these variables integrate traditional physical engineering variables into one overall variable; such variables are not always in common use - hence, the reader may have some difficulty understanding all the implications of such variables until an in-depth reading of the main body of the report is undertaken. For example, such a variable is the "transport rate out of the tailings mass". This is a function of physical variables such as advective groundwater flow (governed by hydraulic gradients and directional permeabilities), diffusion coefficients, and the rate of uptake and dissolution of aqueous soluble forms and particulate forms of chemical elements such as calcium carbonate and sulphate within the tailings mass. Some of these variables are measurable in a fairly straightforward fashion (e.g., the direction of groundwater flow) while others are not (e.g., the rate of change of geochemical mechanisms controlling the uptake-dissolution mechanisms over a 10 000 year time frame). Furthermore, some of the variables which are the least quantifiable are among the most important variables of concern.

The time scale of concern for the types of changes discussed in this document lies between 100 years and 10 000 years. Man is the principal biological organism of concern. The former time frame is one within which it is reasonable to assume that the social system of man might be able to engineer, to manage, to environmentally manipulate and to monitor the biosphere surrounding the disposal site. The latter time scale defines a limit beyond which glaciation might be expected to be a dominant factor controlling tailing contaminant dispersion and dilution. Using effects on human populations as the focus for concern, it is unreasonable to use a time scale longer than that of an ice age since glaciation may force man to leave the area completely. However, considering the technological changes of the last 100 years, this may be a spurious assertion.

For the 100 to 10 000 year time-frame, relatively long-term biogeochemical processes and cycles are of concern (e.g., long-term diffusion rates, lake circulation processes, weathering cycles, the occurrence of earthquakes and severe meteorological events). The initial time of 0 to 100 years is of concern primarily as it relates to the state of existence of the geochemical cycle 100 years after tailings disposal (e.g., physical consolidation of tailings may not be complete and long-term redox boundaries may still be stabilizing).

8.2 PROBLEM DEFINITION

A fundamental series of questions for a lake disposal option are the following: if a contaminant is part of or attached to a non-mobile particulate, is it leachable? If leachable, is it transportable? If leachable, does the leaching rate change, due to a change of chemical mechanisms over time? If transportable, is it removed to some degree to non-mobile particulates along the transport route? If transportable, in which direction is it transported - to surface waters or to deep bedrock waters? If towards surface waters, is the rate of entry into surface waters likely to have a biological impact?

These questions may be simplified to the following fundamental concepts. We are dealing with: (1) a source, (2) pathways from the source, (3) transport mechanisms along these pathways, (4) whether or not the pathways have chemical or biochemical mechanisms which minimize transport (i.e., compensating mechanisms), and (5) whether or not a pathway ends in a biological or chemical form which may adversely affect man or the natural system. The principal contaminant pathway

is outward from the tailings. Diffusion is an example of a transport mechanism. System controllers include the relative importance of physics (e.g., advection and diffusion) versus biochemical and geochemical mechanisms for controlling the rate of uptake and dissolution reactions between soluble aqueous forms and particulate forms. These examples describe which pathways and transport mechanisms are followed by contaminants from the tailings to surface waters or to groundwater. Other examples of pathways are rainfall-runoff, of transport mechanisms are runoff of radionuclides and carbonate from the watershed to a lake, and of system controllers are the weathering reactions driven by atmospheric acidity. These latter examples dictate the background concentration of various materials in the surface waters and how these concentrations change over time.

The study also develops the concept of an "interaction point" between two sets of pathways as a conceptual tool for managers. The "interaction point" between the pathways of leached contaminants from the tailings and pathways from the atmospheric-watershed system meet in the lake water because of biological uptake of contaminants and its transfer up the food chain. The relative increment above background due to the leaching of contaminants from the tailing mass is a prime management variable. However, for some materials such as organic carbon, because the movement of such materials into tailings can cause a change in the biochemical system controlling dissolution in the tailings, the "interaction point" between pathways from the tailings and pathways from the atmosphere is in the tailings mass itself. Accordingly, one should assess the biogeochemical role of the element (e.g., as a pH controller, as an Eh controller, or as an innocuous element such as potassium) and whether there is any particular "interaction point" which should be considered in determining the adequacy of a proponent's evaluation of a lake disposal site.

8.3 MAJOR VARIABLES SELECTED

8.3.1 Description of Major Variables

A list of major variables which should be considered as a minimum list for deep lake disposal is given in Table 8.1. The main basis for the selection of these variables is the degree to which they contribute to transport of leached material, minimize such transport, or control environmental concentrations of various elements. These variables are now described in the order in which they appear in the table rather than in order of their importance.

Table 8.1: Selected major variables

	Relative Importance* of Factor
Tailings - Sediment Systems	
1. Major biochemical and geochemical mechanisms controlling uptake-dissolution interactions within the tailings	1
2. Physics of transport	2
3. Biochemical and geochemical systems in tailings barrier and sediments	1
4. Bleed rate (net outflow rate) of contaminants	3
Hydrogeology	
1. Trend of isostatic rebound	8
2. Physical and chemical stability of soils and bedrock	4
3. Intersection of groundwater aquifers with downstream surface waters	7
Lake Water	
1. Development of biomass	7
2. Bioconcentration effects	7
Lake Water - Sediment Interactions	
1. Stability of and degree of depositional nature of lake (e.g., present day knowledge, fingerprint parts)	4
2. Time of suspension of particulates if eroded	9
3. Sequestering ability and settling rate of lake organic and inorganic particles	7
Lake Water - Sediment - Hydrogeological Interactions	
1. Ratio of transport towards surface water versus towards deeper groundwater	5
2. Assimilative capacity of bedrock	8

Table 8.1: Continued

	Relative Importance* of Factor
Atmosphere - Drainage Basin - Lake Interactions	
1. Long-term stability of lake level	7
2. Frequency of vertical lake overturn	5
3. Long-range stability of atmosphere characteristics	8
4. Long-term input of nutrients and organic matter to lake	7
5. Rate of erosion of particulates from watershed	7
6. Stability of buffering system controlling lake pH	7
7. Background concentration of contaminants of concern	6
Man	
1. Man as a geochemical agent	?

* Relative importance of factor is based upon the degree to which it contributes to the transport of leached material, minimizes such transport, or controls environmental concentrations of important elements. The numbers given are strictly for the purposes of ranking.

Within the tailings mass and lake sediments, the biochemical and geochemical mechanisms which control uptake-dissolution reactions, and their long-term stability, are most important variables over the long term. If there is insignificant erosion of tailings or sediment particles and if no contaminants are leached from the tailings mass, there will be no transport of contaminants, despite favourable aquatic pathways. The long-term stability of these mechanisms must be assured in order to give one confidence that transport will be either negligible or acceptable because of down-gradient attentuation mechanisms. The long-term stability is a function of groundwater and diffusional pathways to the tailings mass, and of the materials that will be carried to the tailings mass along these pathways. If transport of acidic materials occurs, the pore water pH of the tailings-sediment system will be changed. If significant amounts of organic matter are transported into the tailings, bacterial communities may develop which will change the leaching of accessory minerals such as sulfides.

It is important to make measurements of master porewater variables such as pH, Eh, ionic strength, major ions and radionuclides in present lake sediments. These variables are indicators of the chemical character of the porewater and of present biochemical and geochemical systems controlling these master variables. They indicate the potential for transport (actual transport is equal to the product of a concentration term and a physical rate term such as flow rate or diffusivity coefficient). Knowledge of levels of these parameters together with other analyses such as X-ray diffraction for determination of minerals present in existing lake sediments, allow inferences about controlling mechanisms. However, it is the future biochemical and geochemical mechanisms and their behaviour over time which form the fundamental variable controlling potential transport of soluble materials along pathways away from the tailings mass.

Within the tailings, it is expected that formation-dissolution of multicomponent chemical precipitates (e.g., vivianite, gypsum-radium coprecipitates) and adsorption will be the mechanisms most frequently used to express the relationships between soluble concentrations of elements and the mass of each element contained in the particulate phase. Which soluble- particulate relationship dominates at any particular point in time will be governed by the relative importance of the different biochemical and geochemical mechanisms. For example, during initial periods following disposal, calcium sulphate and associated radioisotopes may dissolve providing a potential for transport while during later time periods groundwater inputs of calcium may preclude significant dissolution.

The physics of transport consists of two mechanisms - advection (or convection due to groundwater flow) and diffusion (due to molecular bombardment or eddy processes). Advection is normally described as a flow rate (e.g., m^3/day, a function of the hydraulic gradient and of the parent material's intrinsic permeability) across a cross-sectional area. Diffusion is normally parameterized as a diffusion coefficient (e.g., m^2/day). Transport due to advection will occur only if a hydraulic gradient exists while transport due to diffusion will occur only if a concentration gradient exists.

Next to geochemical mechanisms, the physics of transport is the second most important variable. If the physics of transport is insignificant, then there will be no bleed rate. However, if connected void spaces exist between the tailings mass and surface water, some transport due to diffusion will occur. If the only significant mechanism in the particulates is solid state diffusion, processes in the solid phases would still be more important than pore water transport because solid state diffusion is the rate-limiting step.

Along solute transport pathways through the lake sediments and tailings barriers, biochemical and geochemical mechanisms are deemed to be as important as similar mechanisms in the tailings because they provide a sequestering ability. A sequestering ability can prevent a material from migrating even if it is mobilized and transported from the tailings. It is expected that two major sequestering mechanisms will be formation of coprecipitates (e.g., vivianite) and adsorption (e.g., onto humics, clays, and silts). Ion exchange may be a sequestering mechanism but it may be thought of as a special form of adsorption. Complexation with dissolved species will be a mechanism for keeping contaminants in solution.

The bleed rate out of the sediment-tailings system is the actual transport rate of contaminants out of the sediment-tailings system. The bleed rate is the sum of transport due to advection (the flow rate times the concentration) and due to diffusion (the diffusion coefficient times the concentration gradient times the cross-sectional area) It is a measure of the amount of material which can potentially be carried into surface waters if no further sequestering of contaminant occurs along pathways to surface waters (e.g., by settling organics in a lake water column pathway, or by minerals along a groundwater pathway). It is usually assumed that the bleed rate describes only transport of soluble contaminants. It may have different rates in various directions. For example, material can diffuse in the direction opposite to groundwater flow but net transport up the hydraulic gradient will be smaller than down the hydraulic gradient flow direction if no

differential sequestering by particulates occurs. If groundwater flow is mainly horizontal, vertical transport will be smaller than horizontal transport. Further, the vertical bleed rate may be smaller than a horizontal bleed rate, because of the presence of more sequestering particles (e.g., sediments, a clay barrier) in strata above the tailings.

The relative importance of physical processes versus biochemical and geochemical processes for controlling the bleed rate is conceptually important. If physical transport is relatively fast and if a contaminant solubilizes relatively slowly, then the geochemical rate of release will be the "rate-limiting step" or the mechanism which actually controls the magnitude of bleed rate. On the other hand, if the physics of transport are sufficiently slow that "equilibrium" is maintained between the particulate phase and the aqueous phase, then the physics of transport will be the "rate-limiting step" or the mechanism which controls the magnitude of bleed rate. A transition zone exists where both the physics of transport and geochemical mechanisms control the magnitude of the bleed rate.

Isostatic rebound describes the continuing long-term trend of post-glacial uplift of bedrock. It offers the potential for changing flow patterns within watersheds of some areas of Canada. It may influence the long-term surface level of the lake and could cause changes in outflow locations, depending upon the surrounding topography.

The physical and chemical stability of the soils and bedrock allow the assessment of the ability of a watershed to maintain present physical characteristics as they affect groundwater flow. If a seismic event occurs to fracture bedrock, a significant change in net direction and rate of groundwater flow may occur. If the freeze-thaw cycle further fractures bedrock or if relatively soluble bedrock such as limestone is eroded, the direction and rate of groundwater flow may also change. Changes such as these can significantly alter the magnitude of the bleed rate towards groundwater pathways which intersect with surface waters and hence with the biosphere. Other new pathways (fissures) may direct flow more towards deep bedrock areas. The confidence that a manager has in predictive capabilities for forecasting the future direction of transport must be determined before bedrock instability can be viewed as a potentially beneficial, neutral or detrimental phenomenon.

Intersection of a permeable groundwater zone with down-gradient surface waters indicates a future potential for the physical transport of material bleeding from the tailings to reach surface waters in the

future. In addition, the potential for geochemical retardation along this pathway must also be evaluated.

Within lake water, there are several variables of concern. Two are listed in Table 8.1 as properties of lake water while the rest are listed as properties of lake water interacting with other hydrological units. The mass and species of algae and other biota which develop are a function of external (e.g., solar radiation) and internal (nutrients, pH, etc.) stimuli. The sedimenting of biological particulates and of material eroded from the watershed are mechanisms for the potential removal of contaminants from the water column. Bioaccumulation effects describe the amount of material in different levels of the food chain. The degree of bioaccumulation at different levels in the food chain is a measure of the potential impact upon organisms within these trophic levels. Knowledge of predatory relationships with aquatic organisms (e.g., by bear or man upon fish) and dosimetry and/or toxicity calculations allow estimation of the impact upon various land-based organisms.

At the lake sediment-water column interface, the depositional nature of the lake-sediment system and the associated stability of the sediment is a most important variable for describing whether or not significant transport of sediment-tailings particles will occur. Tailings should be sited only in depositional areas of a lake. The determination of this character is relatively easy (see Section 2.2) if the main deposition features are not changed through the placement of tailings. If, however, the tailings occupy a significant portion of the volume of a shallow lake, some predictive modelling work of lake physics may be essential to permit analyses of changes in depositional features. Examination of existing lake sediments for fingerprint features such as varved sediments, previous mineral history and previous sediment mixing history (e.g., a uniform vertical profile over a few cm which indicates mixing of that depth of sediments) would also aid in ascertaining the long-term depositional features of the lake. Pb-210 dating will frequently be the tool used to estimate the sediment age and to examine their mixing history.

If particulates are eroded from the lake sediments, the time that sediment particles spend in the water column, called herein the "hang-time", is a variable which is useful for assessing their potential ability to release contaminants. If, for example, significant suspension occurs once every 100 years, if the average time of particle suspension is one hour, and if the rate of desorption is relatively slow, it is probable that insignificant quantities will be released compared to long-term transport through porewater pathways.

If the lake is meromictic (exhibits permanent chemical stratification), the buoyancy forces resisting wind-driven vertical mixing act as a significant mechanism for increasing the depositional character of the lake. The benefit of this is a much reduced likelihood of particulate resuspension and extended diffusion distance, and the formation of metal sulfides which are capable of acting as sequestering agents for migrating contaminants.

The sequestering ability and settling rate of lake organic and inorganic particles are measures of the rate of removal of tailings-contaminants from the lake water column. They represent a fundamental mechanism for decreasing the portion of contaminants that will actually reach the more productive lake surface waters. If organic and inorganic particles have an equal sequestering ability, the inorganic particles will be more efficient at removal per unit volume of particle due to their higher settling velocity.

Variables listed in Table 8.1 under the title of lake water-sediment-hydrogeological interactions describe the possible directions that contaminants may follow after release from the tailings. A criterion for site selection may be that preference would be given to contaminant loss directed towards deep bedrock rather than towards surface waters or toward groundwater intersecting with surface waters. Relevant parameters are the assimilative capacity of bedrock and the intrinsic advective rate of flow towards deep bedrock. Beyond a 10 000 year time-scale, it is probable that a large fraction of many radionuclides in the tailings (e.g., Th, with 10-12 half-lives in the million year range) will be liberated due to solid-state diffusion through small particles. The question then becomes whether it is preferable to allow potential migration towards surface waters or towards deep bedrock areas. If the bleed rate is extremely low, this question is moot.

Selection of mitigating measures such as a tailings retaining barrier would influence the predominant direction of contaminant movement. Insertion of a barrier material like clay whose permeability is lower than that of a deeper, fractured bedrock would cause a significant shift in the net direction of groundwater flow. A clay-apatite liner may preferentially absorb sufficient amounts of contaminants (e.g., radium or uranium) such that it decreases the total bleed rate from the sediment-tailings system. Meromictic waters may favour greater consolidation of sediments, thus decreasing the rate of transport through sediments overlying tailings.

Variables described as atmosphere-drainage basin-lake interactions describe the mixing characteristics of a lake and the long-term ability of the system to maintain existing chemical systems (e.g., buffering systems controlling the lake's pH, nutrient regimes, and background levels of contaminants). The long-term stability of the lake level is inversely related to the probability that the lake may become sufficiently shallow that depositional sediments become erosional sediments or that it will dry up, exposing dry sediments to wind-driven erosion and possibly to alternating redox regimes.

The frequency of vertical lake overturn gives an indication of the rate of vertical transport in the lake once the contaminant enters the water column and of whether advection or diffusion will dominate lake transport. In shallow Canadian lakes, overturn usually occurs once or twice per year. In deep lakes protected from high winds, overturn may occur only once every decade; however for a 10 000 year time-frame, such events are sufficiently frequent that the lake approaches well-mixed conditions. If artificially-induced or natural meromixis prevents mixing from occurring for long time periods, diffusion may dominate.

For the long-term maintenance of present water chemistry, the first variable of importance is the maintenance of present atmospheric characteristics. This may be divided into globally controlled properties (e.g., wind, rainfall, etc.) and culturally modified properties (e.g., atmospheric chemistry including acidity). Global properties will change in unforeseeable ways, inducing such lake responses as lower water levels due to drought. These are difficult to assess except by fingerprint variables such as past cultural history (e.g., Lake Geneva, see Section 6.4) and sediment variables (e.g., changes in formerly dry, nearshore sediments). Margins of safety in design are needed to handle such unpredictable variations. For man-induced changes, scenario testing (what if's) or design ranges (selected by relevant authorities) are two approaches for estimating potential changes.

The long-term inputs of nutrients and organic matter to a lake maintains the lake's ability to use sunlight energy for the growth of biological particles and ultimately to form new lake sediments. These inputs result primarily from watershed processes which are controlled by weathering reactions and terrestrial productivity. Significant inputs of nutrients directly from the atmosphere to a lake occur in some lakes. Soluble forms of these organics can migrate into tailings provided suitable pathways exist; the potential for these organics to

control the biochemical mechanisms of the sediment is dependent upon the quantity of transport and the bioavailability of the organics, particularly under "reduced" conditions.

The rate of erosion of particulates in the watershed gives an estimate of the rate of burying of sediments and hence of the rate of segregating of the tailings layer from the water column. It is a measure of the potential for sequestering trace metals or radionuclides in the water column and removing them through settling to the sediment.

The stability of a buffering system controlling the lake's pH strongly influences the biota of the lake and the form of materials which diffuse into the sediment-tailings system. It is controlled by inputs of alkalinity and total inorganic carbon in most lakes. These inputs are influenced by weathering reactions and rainfall acidity.

The background environmental concentrations of radionuclides and other contaminants before tailings disposal provide the baseline against which an acceptable bleed rate is measured. For example, a bleed rate equivalent to 2 pCi/L in lake water may be acceptable relative to a background concentration of 10 pCi/L, but not acceptable relative to a background of 0.1 pCi/L. The background concentration results from atmospheric inputs directly to the lake and from atmospherically-induced weathering of the watershed.

8.3.2 Relative Importance of Major Variables

The relative generic importance of the various variables is suggested in Table 8.1. The order of these variables is in some instances arbitrary. The degrees of difference between the relative importance of variables cannot be determined in a general sense, because these differences will be site-specific. The actual order of importance may change depending upon the lake and watershed in question.

The main point determining the order of these variables is that the first group of mechanisms is concerned with whether contaminants will or will not enter a soluble phase, whether the geochemical and biochemical mechanisms controlling this leaching changes over 10 000 years, and whether or not the contaminant will be transported. The variables on the next level of importance (4-5) describe the pathways to surface waters, their long-term transport character and their disturbances. The next level of variables (7-8) describes the lake's ability to sequester contaminants from its watercolumn and controls on

lake water chemistry, including controls in "background concentrations". The last level of variables describes transport caused by the movement of tailings particles; their relative importance will be quite site-specific.

One variable listed, but not described nor assessed is "man". Given the past trends of human civilization and its ability to alter the surface environment, man can be expected to impact the lake-tailings system in the future. However, the exact direction cannot be predicted with certainty. Using past history as a guide, man has tended to speed up transport of material along various pathways. In a lake system, it may be expected that his unconscious behaviour will speed up such processes as erosion and atmospherically-induced weathering reactions. Design and site evaluation should consider these potential effects. Man has the potential for directly upsetting the lake-tailings system. Analogies to present day archaelogy and deep-sea diving suggest that direct disturbances are quite feasible now and over the long term. Whether or not it is reasonable to anticipate and to plan for such behaviour or merely to provide safety factors in the design of a tailings-system awaits determination.

8.4 SUMMARY OF ALL VARIABLES CONSIDERED

The placement of tailings in a lake system presents many environmental concerns which are difficult to assess. Material presented in this document is fairly comprehensive. All the major factors which are discussed in the main body of this report and possible methods for measuring them, are summarized in Table 8.2. Examination of this table allows one to obtain a general comprehension of the number of variables and possible measurements which are relevant to consideration of the long-term containment question. One can see "how the transporting hydrological units fit together", but the thread is still potentially difficult to discern because of the large degree of connection between adjoining hydrological units. Further, for some potential sites, a proponent may be able to combine several hydrological units together and consider them as one entity rather than several entities. It is probable that many of the factors considered in this report will be inconsequential at some sites and may be neglected.

Table 8.2: Summary of variables and possible measurements of these variables for all hydrological units

1.0 SEDIMENT-TAILING SYSTEM

1.1 U-MINE TAILINGS IN SEDIMENTS

1.1.1 Biogeochemical Characteristics of Tailings

1.1.1.1 Chemical form of tailings upon deposition

Possible Measurements

1. percent of each contaminant in different forms
2. rate of breakdown or dissolution of particulate forms

1.1.1.2 Geochemical Mechanisms

Possible Measurements

1. pore water concentrations
2. crystalline forms
3. uptake-dissolution mechanisms
4. temperature

1.1.1.3 Biochemical Mechanisms

Possible Measurements

1. composition of organic matter
2. probable biological communities
3. possible chemical leaching agents
4. temperature

Table 8.2: Continued

1.1.2 Physical Characteristics of Tailings

1.1.2.1 Initial Characteristics upon Deposition

Possible Measurements

1. particle-size distribution
2. density
3. organic-inorganic ratio
4. settleability
5. consolidation rate

1.1.2.2 Porewater Transport Characteristics

Possible Measurements

1. vertical and horizontal permeability coefficient
2. diffusivity coefficient

1.1.2.3 Stability and Cohesiveness

Possible Measurements

1. stability tests in laboratory
2. cohesion tests in laboratory

1.1.2.4 Solids Transport Processes

Possible Measurements

1. consolidation processes
2. solid diffusion coefficient

Table 8.2: Continued

1.2 TAILINGS BARRIER

1.2.1 Physical Characteristics

1.2.1.1 Consolidation Processes

Possible Measurement

1. long-term consolidation rates

1.2.1.2 Stability

Possible Measurements

1. cohesiveness
2. ability to resist seismic activity

1.2.1.3 Porewater Transport Characteristics

Possible Measurements

1. permeability coefficient
2. diffusivity coefficient

1.2.2 Chemical Characteristics

1.2.2.1 Uptake Characteristics

Possible Measurements

1. chemical nature of barrier material
2. uptake mechanisms

1.2.2.2 Stability

Possible Measurement

1. consolidation characteristics of clay mixtures

Table 8.2: Continued

1.3 LAKE SEDIMENTS

1.3.1 Lake Sediment Physics

1.3.1.1 Consolidation Processes

Possible Measurements

1. long-term consolidation rates of present lake sediment
2. consolidation factor in sediment transport models

1.3.1.2 Transport Mechanism Through Porewater

Possible Measurements

1. vertical advection
2. vertical diffusion

1.3.1.3 Stability

Possible Measurements

1. ability to resist shear forces (e.g., shear tests)
2. macroinvertebrate densities (bioperturbation effects versus depth of habitat zone)

1.3.1.4 Accumulation Characteristics

Possible Measurements

1. depositional or erosional nature of environment
2. ratio of sediment accumulation rate to lake sedimentation rate
3. fingerprint features of existing sediments

Table 8.2: Continued

1.3.2 Lake Sediment Biogeochemistry

1.3.2.1 Geochemical Measurements

1. correlations between contaminants and physical or chemical sediment variables
2. correlations between porewater elements and particulate phase parameters
3. minerals observed in sediments
4. porewater concentrations of elements involved in diagenetic processes

1.3.2.2 Biochemical Processes

Possible Measurements

1. percent organics
2. redox potential, O_2, S^{2-}
3. dissolved organics in porewater
4. source of sediment organics
5. stability of sediment organics (biodecomposability, sediment oxygen demand)

1.3.2.3 Biological Communities

Possible Measurements

1. major bacterial communities and numbers
2. macroinvertebrate characteristics and densities
3. measures of bulk community properties

1.3.3. Relative Importance of Transport and Biogeochemical Mechanisms

Possible Measurement

1. dimensionless numbers of biogeochemical and transport variables

Table 8.2: Continued

2.0 LAKE WATER BODY

2.1 LAKE WATER RELATIONSHIPS

2.1.1 Water Body Physics

2.1.1.1 Vertical and Horizontal Lake Characteristics

Possible Measurements

1. horizontal versus vertical character
2. direction of chemical gradients (present and future)

2.1.1.2 Vertical Circulation Characteristics

Possible Measurements

1. frequency of vertical overturn
2. maximum depth of vertical overturn
3. rate of thermocline erosion during summer stratification
4. stability of stratification (e.g., Richardson number)
5. depth of outlet sill

2.1.1.3 Horizontal Circulation Characteristics

Possible Measurements

1. dominant horizontal currents
2. exchange rates between embayments and lakes
3. time-scale of achieving well-mixed conditions
4. chemical horizontal gradients

Table 8.2: Continued

2.1.2 Water Body Chemistry

2.1.2.1 Chemical Characteristics

Possible Measurements

1. buffering system controlling the pH
2. oxygenation characteristics
3. POC and DOC characteristics
4. major cations and anions
5. background radioisotope regimes

2.1.2.2 Relevant Biological Growth Factors

Possible Measurements

1. nutrient (N, P) regimes
2. essential trace metals
3. vitamins
4. toxic trace metals and controlling organic compounds

2.1.3 Water Body Biology

2.1.3.1 Biological Components

Possible Measurements of Phytoplankton and Ecological Significance

1. species composition in water
2. species composition in sediment cores
3. biomass
4. primary productivity
5. particle size distribution

Possible Measurements of Macrophytes and Ecological Significance

1. areal extent of littoral zones
2. species composition
3. potential for macrophyte grazing by animals

Table 8.2: Continued

Possible Measurements of Zooplankton and Ecological Significance

1. species composition in the water column
2. species composition in sediment cores
3. density/biomass in the water column
4. potential for food chain transfer
5. particle-size distribution

Possible Measurements of Benthic Macroinvertebrates and Ecological Significance

1. species composition
2. species composition in sediment cores
3. density or biomass
4. bioperturbation potential
5. depth of occurrence in sediments

2.1.3.2 Diversity and Stability of Ecosystems

Possible Measurements

1. Shannon-Weaver diversity index
2. Brillouin diversity index
3. production/biomass ratio

2.1.4 Water Body Inorganics

2.1.4.1 Physical Characteristics

Possible Measurements

1. settling rates
2. size distribution and mineralogy

Table 8.2: Continued

2.1.4.2 Chemical Characteristics

Possible Measurements

1. sources of inorganics
2. adsorption characteristics

2.1.5.1 Biological production characteristcs

Possible Measurements

1. standing crop of algae, zooplankton, rooted aquatics
2. rate of production of various plants
3. rate of secondary production
4. potential for food chain transfer

2.1.5.2 Biological Decomposition Characteristics

Possible Measurements

1. bacterial and fungal populations present
2. ATP concentrations
3. rates of oxygen consumption in the water column

2.1.5.3 Ecological Relationships of Community Structure to Water Chemistry

Possible Measurements

1. sediment fingerprints due to previous changes in lake ecosystem
2. case studies from other lakes

Table 8.2: Continued

2.1.5.4 Other uptake mechanisms

Possible Measurements

1. adsorption isotherms
2. complexing capacity
3. potential precipitates and properties

2.1.5.5 Other Dissolution Mechanisms

Possible Measurements

1. leaching measurements
2. redox reactions in anoxic water

2.2 LAKE SEDIMENT-WATER INTERACTIONS

2.2.1 Lake Influences on Sediment Properties

Possible Measurements

1. zones of accumulation and erosion
2. water content of sediments

2.2.2 Net Removal Mechanisms of Trace Metals and Radioisotopes from Lake Water Columns

Possible Measurements

1. amount of ionic precipitation
2. amount of biogenic precipitation
3. amount of complexed and adsorbed metals
4. settling velocity of the different particulate fractions

Table 8.2: Continued

2.2.3 Sediment-Water Transport

Possible Measurements

1. present concentration gradients
2. net vertical transport
3. vertical particle movement (particle "hang-time")
4. horizontal erosion
5. biological grazing patterns

2.2.4 Perspective on Lake Sediment-Water Interactions

3.0 HYDROGEOLOGY

3.1 PHYSICAL STABILITY OF BEDROCK

Possible Measurements

1. type of bedrock
2. presence of major faults
3. history of seismic activity
4. historical uplift character of plate
5. fractured nature of bedrock
6. long-term maintenance of fracture characteristics

3.2 CHEMICAL STABILITY OF BEDROCK

Possible Measurements

1. dissolution characteristics of bedrock for different aggressive waters
2. potential precipitation reactions from groundwater

Table 8.2: Continued

3.3 GROUNDWATER FLOW CHARACTERISTICS

Possible Measurements

1. porous media versus fracture media characteristics
2. directional permeabilities
3. bulk diffusivity coefficients
4. fingerprint features

3.4 BIOCHEMISTRY OF GROUNDWATER

Possible Measurements

1. organic characteristics of recharge areas
2. potential for oxygen migration

4.0 INTERACTIONS BETWEEN TAILINGS, LAKE SEDIMENTS AND HYDROGEOLOGY

Possible Measurements

1. advection flow direction
2. fingerprint features for advection
3. advective and diffusive transport direction
4. flux of chemical elements into tailings
5. flux of chemical elements out of tailings
6. probable long-term geochemical and biochemical mechanisms
7. detention of materials in sediment-tailings systems
8. susceptibility of receiving waters to these fluxes

Table 8.2: Continued

5.0 DRAINAGE BASIN

5.1 TERRESTRIAL CHARACTERISTICS

5.1.1 Erosional Nature of Watershed

Possible Measurements

1. various parameters in universal soil loss equation (USLE)
   a) volume of runoff
   b) soil erodibility factor
   c) slope, length and gradient factor
   d) crop management factor
   e) erosion control practice factor

5.1.2 Surface Runoff Potential

Possible Measurements

1. various parameters in USLE
2. precipitation-runoff relationships
3. runoff coefficient

5.1.3 Chemistry of Surface Runoff

Possible Measurements

1. major ions in surface water runoff
2. leachate from major bedrock outcrops
3. leachate from major soils

Table 8.2: Continued

5.1.4 Terrestrial Productivity

Possible Measurements

1. gross biomass production
2. biomass production of particular plants
3. area of ecosystem based upon dominant plant communities
4. influence on weathering reactions
5. net organic export to lake system

5.2 DRAINAGE BASIN-LAKE INTERACTIONS

5.2.1 Hydrological Budget Aspects

Possible Measurements

1. average annual lake inflow
2. high-flow extrema
3. low-flow extrema

5.2.2 Material Budget Aspects

Possible Measurements

1. average annual input to lake (e.g., kg/yr.)
2. areal lake loading (e.g., gm/m^2-yr.)
3. lake loading-concentration models

5.2.3 Material Budget Relationships to Biological Regimes

Possible Measurement

1. lake loading-biomass models

Table 8.2: Continued

5.2.4 Transport Upset Due to Extrema

Possible Measurements

1. fingerprint features in lake sediments
2. seismic activity

6.0 ATMOSPHERE

6.1 ATMOSPHERIC CHARACTERISTICS

6.1.1 Radiation Characteristics

Possible Measurements at Study Site

1. solar radiation incident upon earth
2. solar radiation adsorbed by atmosphere
3. atmospheric radiation
4. atmospheric emissivity

6.1.2 Wind Characteristics

Possible Measurements

1. frequency diagram of wind speed
2. frequency diagram of wind direction
3. frequency of occurrence of extrema
4. change in global circulation patterns

6.1.3 Precipitation Characteristics

Possible Measurements

1. rainfall
2. fingerprint features as indicators of past rainfall
3. possible future changes in rainfall quantitites

Table 8.2: Continued

6.1.4 Thermal Characteristics

Possible Measurements

1. temperature
2. future changes in heat fluxes

6.1.5 Atmospheric Chemistry

Possible Measurements

1. relative composition of major gases (e.g., O_2, N_2)
2. pH of rainfall
3. major metals, anions and nutrients
4. nature of particulates

6.1.6 Prediction of Climatic Change

Possible Measurements

1. knowledge of controls on past climatic trends
2. reliability of future predictions

6.2 ATMOSPHERIC INTERACTIONS WITH DRAINAGE BASIN

6.2.1 Hydrological Budget

Possible Measurements

1. average annual rainfall rate
2. average annual evapotranspiration
3. intensity-duration-frequency curves for rainfall
4. average annual runoff
5. statistics of runoff

Table 8.2: Continued

6.2.2 Material Budget

Possible Measurements

1. quantity of elements deposited on drainage basin (gm/ha-yr.)
2. quantity of material removed from drainage basin (gm/ha-yr.)

6.2.3 Atmospheric Perturbations on Drainage Basin

Possible Measurements

1. history of droughts
2. history of flash-floods
3. history of natural forest fires
4. estimates of future rainfall chemistry

6.3 ATMOSPHERIC INTERACTIONS WITH LAKE

6.3.1 Lake Heat Budget

Possible Measurements

1. annual cycle of equilibrium temperature
2. various heat fluxes

6.3.2 Solar Radiation - Lake Interactions

Possible Measurements

1. percentage of short-wave radiation adsorbed at lake surface
2. net extinction coefficient
3. compensation depth

Table 8.2: Continued

6.3.3 Wind Shear Stress - Lake Current Relationships

Measurements

1. wind shear stress
2. empirical wind velocity - lake current formulae

6.3.4 Water Budget and Mass Budget

Possible Measurements

1. ratio of rainfall to evaporation
2. atmospheric deposition rate of particulates
3. atmospheric deposition rate of chemical constituents

6.4 ATMOSPHERIC INTERACTIONS WITH DRAINAGE BASIN - LAKE SYSTEM

6.4.1 Geochemical Weathering Reactions

Possible Measurements

1. watershed material budgets
2. shifts in watershed material budgets

6.4.2 long-Term Trends in Lake Chemistry

Possible Measurements

1. ratio of drainage basin area to lake surface area
2. change in watershed material budgets

Table 8.2: Continued

6.4.3 Erosional Processes

Possible Measurements

1. suspended solids and bedload export from watershed
2. suspended solids - flow rate relationships
3. rate of recession of shoreline bluffs
4. sedimentation rates in a lake
5. depth of groundwater relative to the frost line

6.4.4 Surface Runoff Characteristics

Possible Mesurements

1. characteristics of runoff from watershed
2. characteristics of runoff from lake
3. lake level

6.4.5 Lake Mixing Characteristics

Measurements

1. influence of inflow characteristics upon lake mixing
2. disturbance of lake sediments by extreme currents

6.4.6 Transport from Lake to Watershed Components

Possible Measurements

1. exhalation of gaseous radionuclides from lake
2. possible routes for transport of radionuclides to land-based animals.

Table 8.2: Continued

7.0 HUMAN INTERACTIONS

7.1.1 Human Effects Upon Atmosphere

7.1.2 Human Effects upon Drainage Basin

7.1.3 Human Effects on Lake

7.1.4 Tailings-Barrier-Sediment System

7.2 MITIGATING MEASURES

7.2.1 Engineering of Lake Mitigating Measures

Possible Methods

1. addition of cementing agents to tailings
2. method tailings placement
3. creation of barrier around tailings
4. creation of meromixis
5. bypassing inflowing streams to outflow
6. creation of artificial hills around lake
7. planting trees nearby

7.2.2 Environmental Engineering Mitigating Measures

Possible Methods

1. change land use patterns of drainage basin
2. add nutrients to a lake
3. neutralize watershed or lake
4. meromixis

7.2.3 Natural and Environmental Controls or Meromixis

7.2.3.1 Natural Control of Meromixis

7.2.3.2 Engineered Creation of Meromixis

8.5 ILLUSTRATION OF AN APPROACH FOR EXAMINING THE ACCEPTABILITY OF TAILINGS DISPOSAL IN LAKE SEDIMENTS

A hypothetical example is described in Table 8.3 to illustrate how these factors may be used to evaluate a proposal for tailings disposal. The example considers that tailings are in a headwater lake system, situated in a rather hilly area. Surface inflows and outflows occur to and from the lake but the groundwater inflow to the tailings is only from an adjoining watershed. A simplification of the tailings mass is assumed and described in Table 8.3 (e.g., a radium barium sulfate precipitate, gypsum as the only crystalline material resulting from chemical neutralization and tailings particles containing only i) Ra associated with an accessory sulfide mineral, ii) U in the host matrix, and iii) an arsenide mineral whose composition is unspecified). Then, probable geochemical and biochemical mechanisms and transport mechanisms are traced from the tailings to surface waters (through groundwater and surface water routes).

This analysis presently one possible approach to assessing the long-term processes which would exist in this lake. It allows formulation of engineering variables and management variables. Only a few of these variables given in Table 8.2 are significant because geological upset in it is not probable, it has excellent depositional features, it is deep and sheltered, and the tailings volume is small.

In addition to transport of soluble phases, a prime fear for lake disposal is the erosional character of the bottom waters (due to a meteorological extrema disturbing the tailings). In this example, the lake has been assessed as having non-erosive bottom waters - even if subjected to a hurricane force wind. If a lake has erosive characteristics under meteorological extrema, analyses of the "hang-time" of the particles and the subsequent amount of desorption is an additional variable which should be determined.

This analysis allows one to present a decision in terms of the net increment above background that the long-term leaching of tailings contaminants will cause in the lake's surface waters and in downstream waters. The acceptability of such a focus for decision criteria probably needs to be debated in Canadian Society and perhaps altered, if appropriate. Some form of decision criteria or goals may need to be endorsed by regulatory authorities in order to aid a proponent

in their analysis. Formulation of goals rather than of criteria would arise from the understanding that sufficient knowledge may not, at present, be available concerning certain geochemical mechanisms, although this knowledge may become available in the near future.

The analysis also gives estimates of which management and engineering variables are practical and which are more of a research nature at present.

Table 8.3: Example analysis of tailings placed in lake

I Tailings System

1. Description of Tailings (Note: the simple tailings system illustrated in Figure 0.1(b) is used here)

- The tailings consist of 3 forms
  - a radium barium sulfate ($RaBaSO_4$) amorphous precipitate
  - a host tailings particle which has some U in the matrix and an Arsenide mineral
  - a calcium sulfate ($CaSO_4$) precipitate, which enmeshes some Th and which covers part of the surface of the host tailings particle

1a. Transport Pathways

- The net groundwater gradients are downward to deep bedrock with a (vertical permeability of 10^{-6} cm/s. A small aquifer flows horizontally to a downstream surface water (horizontal permeability coefficient 10^{-8} cm/sec).

- The main groundwater pathway is an aquifer which connects a marsh from an adjoining watershed directly to the tailings. Its chemistry is dominated by organics, calcium and total inorganic carbon (TIC).

Possible geochemical and biochemical factors in the tailings are as follows:

- If they are anaerobic, the $RaBaSO_4$ and $CaSO_4$, formed under oxidizing conditions, will have a tendency to dissolve.
- High $[SO_4^{-2}]$ will probably favour Th transport (due to chemical complex formation) but not Ra transport; however, if they are anaerobic, sulfate would be reduced sulfide. Then the question is whether the thorium sulfate ($ThSO_4$) complex remains intact.
- If NO_3 is initially present, it may prevent $SO^2{}_4^-$ reduction; however, it would probably be reduced during the first 100 years, allowing $SO^2{}_4^-$ reduction to commence.

Table 8.3: Continued

2a. Estimate of Probable Dominant Mechanisms and Systems in Tailings

- pH will initially be basic due to high initial concentrations of $CaCO_3$ and due to the groundwater being high in $CaCO_3$. It will be pushed towards the acidic range, depending upon the amount of anaerobic decomposition of organics, sulfide concentration, and TIC. The TIC system will probably control pH, although sulfide may dominate for a particular time.
- Eh will be between -100 and -400 mV depending upon the rate of organic carbon breakdown.
- Some FeS minerals and associated radionuclides will dissolve following natural geochemical pathways, but the rates will be slow relative to aerobic conditions. Aerobic conditions increase solubilization of accessory minerals by several orders of magnitude due to conditions favourable for growth of bacteria which feed upon accessory minerals.
- Biochemical processes (anaerobic decay, fermentation of organics) may increase the rate of FeS and arsenide dissolution, although the literature is mixed on this point. The presence of organics is inevitable; whether it is desirable to insert organics into the tailings is unclear. Since the minerals were formed originally (several million years ago) under anaerobic (reduced) conditions, maintenance of such conditions is a useful rule of thumb. Dissolution will occur geochemically if pore water concentrations are less than saturation depending upon the resultant mineralogy. Anaerobic processes may increase or decrease this rate of dissolution. The presence of appropriate organics, if complexed with U, will increase its soluble concentration, increasing the driving force for transport.
- The amount of $RaBaSO_4$ remaining as particles will be controlled by balance between the dissolution of sulfate and barium and diagensis, (which changes its structure from amorphorus to more crystalline and effectively increases the solubility product). Ra dissolution could still occur due to solid-state diffusion.
- Compensating mechanisms will exist. Formation of vivianite (FeS) from super saturated pore water is possible. It may coprecipitate some metals and almost certianly surface sorb others.
- U^{+4} will be reduced to U^{+2} and demobilized.
- $CaSO_4$ (and Th) dissolution will be slow due to presence of Ca in groundwater inflow.

Table 8.3: Continued

2b. Estimates of Probable Physics of Transport

- The milling process has changed the solid state diffusion coefficient from being one number to being a range of numbers (10^{-7} to 10^{-12} cm^2/sec) depending upon the nature of the phase containing the contaminant (e.g., pore water, amorphorus precipitate, host-rock).
- The diffusivity coefficient in the pore water may be equal in all directions.
- The dominant groundwater flow will be downward, with a very slow rate horizontally to the surface aquifer. The groundwater flow will be to an aquifer at 200 m which, it is estimated, connects to surface waters (due to advective flow) in 5 000 yr. and over a linear pathlength of 5 km.
- Horizontal and vertical flows will be determined by the regional hydraulic gradient of the system (drainage basin-lake-sediment-tailings-groundwater) and the directional permeabilities.

3. Suggested Approach to Analysis of Tailings is a Mass Balance Analysis

1. Use mass balance for transport of organics into and out of tailings SO_4^{2-} NO_3^-, and O_2 to estimate Eh.
2. Use mass balance on alkalinity and TIC into and out of tailings and reactions within the tailings to estimate pH.
3. Estimate controlling biochemical and geochemical mechanisms.
4. Use equilibrium chemistry models (e.g., Morel et al, 1975) to estimate ionic strength, individual pore water species, amount adsorbed, and the quantity precipitated; use mass balance of S, N, O_2, organics and inorganic ions to check dominant geochemical and biochemial mechanisms, and whether physics of transport is the rate limiting step.
5. Set up a table of "sums" which describe change of mass of different materials in the tailings as a function of time. This provides a check on when a biochemical or geochemical system will change due to the depletion of one material (e.g., if all readily available sulfide has been dissolved from sulfate and sulfide minerals, a shift of system will occur).

Note: In the mass balance analysis, the hydraulic gradient, the permeability coefficient and the diffusion coefficient together with the upstream and downstream concentrations need to be known. These may be initially calculated assuming that the system is at steady state.

Table 8.3: Continued

4a. Conceptual Model for Describing Transport (the example material considered here is sulfide)

- Downward bleed rate of sulfide (units gm/yr.)

$$= q_4C_4 + D.A_4(C_5-C_4).L/2$$

where q and C are flows and concentrations as indicated in Figure 0.1 and A_4 is the cross-sectional area of pore water on the bottom of the tailings.

- Estimate of C_5
  - C_5 is a function of the dissolution rate of sulfide minerals + the dissolution rate of sulfate precipitates + net loss of sulfide due to advection and diffusion + net loss to vivianite phase.
  - Simplification: assume that pore water concentration C_5 is controlled by the precipitation reaction with vivianite. Then

$$C_5 = K_{FeS} / [Fe^{2+}]$$

where K_{FeS} is the solubility product of vivianite and $[Fe^{2+}]$ is the pore water concentration of ferrous iron.
Then, i) the downward bleed rate =

$$q_4C_4 + D.A_4 (K_{FeS}/[Fe^{2+}] - C_4).L/2$$

ii) the net loss to vivianite phase = dissolution of accessory sulfide minerals + dissolution rate of sulfate precipitates - net loss due to transport.

- In this simplification, the term $K_{FeS}/[Fe^{2+}]$ is analogous to a distribution coefficient, but one which involves a precipitate as the phase rather than an adsorbed material.

4b. Key points resulting from this conceptual model:

i) The concentration is a function of bleed rate; the bleed rate is a function of concentration. The tautology can only be resolved by a mathematical model, unless some simplifying assumptions can be made.

Table 8.3: Continued

ii) Implicit assumptions made should be stated. For example, the dissolution rate of sulfate from gypsum is assumed to control pH, sulfide is assumed to be formed rapidly from sulfate, under reduced conditions.

iii) The concentration of sulfide is a function of two dissolution rates, that of sulfide minerals and that of sulfate precipitates. The rates may not be well known.

iv) Other precipitates besides vivianite may form.

4c. Conceptual analogy

- The groundwater flow through several strata cannot be calculated until the hydraulic gradient across each stratum is known, even if the directional permeability of each stratum is known. Only when the whole system is put together into a mathematical groundwater model are the individual hydraulic gradients and system flow rates calculated - they are calculated at the same time. Similarly, with the transport problem, individual permeabilities and diffusivity coefficients are known; however, not until the system is put together into mathematical model are the concentration gradients and system transport rates calculated (at the same time). The major difference is that, at present, calculation of chemical gradients is more difficult because of the competing biochemical and geochemical mechanisms present.
- To measure permeability coefficients in groundwater flow, either a pump test is conducted and a permeability inferred or a sample is brought into the lab and permeability is measured (to within an order of magnitude). To measure concentration characteristics of tailings in the field is difficult; leaching tests under different pH and Eh conditions at low flow rates permit macro measurements while knowledge of the characteristics of surfaces of particles and equilibrium chemistry permit more fundamental estimates.

Table 8.3: Continued

- To estimate the change in permeability coefficent and hence the change in groundwater gradients over time, the rate of dissolution of bedrock and soils is necessary. To estimate the change in biochemical or geochemical systems over time, the change in dominant systems using mass balance analysis is required.
- In groundwater flow analysis, the approach is use of a flow balance; in mass transport analysis, the approach is use of a mass balance.

5. Management Variables

1. Residence time of various elements in tailings mass (partially practical)
2. Hydraulic Flow Rate (practical)
3. Mass Transport Rate (partially practical)

Note: Mass Transport rate defines the directional bleed rates.

6. Engineering Variables

1. Directional permeability coefficients (a practical variable)
2. Directional Hydraulic gradients (a practical variable)
3. Diffusivity Coefficient (a practical variable)
4. Chemical precipitation solubility products for multi-component species (mainly a research variable)
5. Chemical adsorption coefficients (a practical variable)
6. Chemical complexation constants for pore water (a practical variable)
7. Change in permeability and diffusivity coefficients (practical)
8. Change in biochemical and geochemical systems (partially practical, but the dominant engineering variable)

Table 8.3: Continued

II Tailings Barrier

1. Description of Barrier
   - Clay liner, 1 m thick, undergoing consolidation and with apatite added.
2. Probable geochemical/biochemical factors
   - dominant mechanisms may be adsorption onto clay, and coprecipitation to form vivianite
   - Eh will probably remain reduced, pH slightly elevated above that of tailings
3. Probable transport mechanisms
   - diffusion may dominate, unless groundwater flow into tailings moves upward in a substantive fashion (see Section 1.2)
   - top particles of tailings barrier may be erodable during first 1 000 years by bottom lake currents (see Section 2.2, 6.4)
4. Suggested analysis approach
   - mass balance analysis of various elements to determine
     i) Eh
     ii) pH
     iii) estimate biochemical and geochemical systems
     iv) estimate pore water concentrations and check systems of iii)
     v) estimate vertical bleed rate
5. Management variables
   1. bleed rate (partially practical)
6. Engineering variables
   - same as tailings except here controls on pH and biochemical systems will be more influenced by lake water.

III Hydrogeology

1. Description of hydrogeology
   - the several strata of bedrock as given in tailings's description
   - bedrock is igneous rock; no limestone
2. Probable geochemical/biochemical factors
   - dominant mechanisms - adsorption onto in-fracture surfaces and coprecipitation to form vivianite. Some diffusion from fracture into rock matrix may occur with resultant long-term contaminant sequestering.
   - Eh and pH will probably remain similar to that of tailings (Note: a proper mass balance analysis would examine this question)

Table 8.3: Continued

3. Probable transport mechanisms
   - advection may dominate over diffusion
   - geotechnical data suggest that bedrock should remain stable for 10 000 yr - no change in transport and character through bedrock is predicted; outflow sill of lake will change only a few inches
4. Suggested analysis approach
   - same as for tailings barrier
5. Management variables
   1. Bleed rate to horizontal aquifer at its intersection with surface waters and the time of breakthrough
   2. Expected accumulation of radioisotopes in deep bedrock over time and their rate of breakdown (e.g., what is the concentration of Th in deeprock at 10 000 yr.)
6. Engineering variables
   - same as tailings, except controls on Eh and pH will be dominated by geochemical processes

IV Sediment System

1. Description of Sediment
   - a slowly accumulating sediment is expected, composed of 20% quartz (from lake's watershed) and the remainder as organics (60% humics from the watershed and 20% algal detritus)
   - for assessment, see Section 2.1.3-2.1.5, 2.2, 5.2, and 6.4
2. Other assessments
   - similar to tailings barrier, except i) adsorption may be a dominant uptake-release mechanism, ii) the sediment chemical system will be more highly influenced by transport of material from the lake water than is the tailings barrier, iii) diffusion and consolidation induced advective transport will dominate pore water transport, and iv) lake sediments may be periodically mixed unless meromixis exists

V Present Lake Water System

1. Description
   - the major buffering system will be the TIC system. The hypolimnion remains aerobic, but meromixis is being contemplated. The nutrient regimes suggest algal dominance by diatoms.

Table 8.3: Continued

2. Probable geochemical/biochemical factors
   - dominant sequestering mechanisms are adsorption onto diatom shells, quartz particles and complexation with humic acids.
   - pH is neutral, Eh is highly oxic but will be reduced in a bottom meromictic layer if such is created (see Section 7.2)
3. Probable transport mechansism
   - advection will dominante - the lake will be well-mixed on the average each year
4. Suggested Analysis Approach
   - mass budget; Eh analysis is not necessary, but O_2 analysis is.
   - mass input-biomass relationships (see Section 2.2) are needed to estimate diatom characteristics
5. Management variables
   1. Contaminant concentrations resulting from tailings (practical)
   2. Background contaminant concentration (mainly practical)
   3. Outflow concentration of contaminant (practical)
6. Engineering variables
   1. Hydrological budget over time (partly research)
   2. Mass balance on lake water of various elements (practical)

VI Present Watershed

1. Description of watershed
   - steeply sloping with 40% tree cover
2. Probable geochemical/biochemical mechanisms
   - weathering reactions and their modification by terrestrial productivity
3. Probable transport mechanisms
   - granite and soil particles will erode; some humics will be exported
   - all runoff is all surface runoff; there is no groundwater or upstream lakes
4. Suggested analyses approach
   - mass balance approach on watershed relating rainfall inputs to watershed export for various elements
5. Management variable
   - Atmospheric input-watershed export relationships (practical)

(see Sections 5.1, 5.2, 6.2)

6. Engineering variables
   - same as management variable (practical)

Table 8.3: Continued

VII Atmosphere

1. Description of atmosphere
   - future changes in climate
   - future changes in CO_2, acidity
2. Suggested analysis approach
   - conduct studies to determine long-term (10 000 yr.) acidity using fuel-burning scenarios (see Section 6.1)
   - use constraints mandated by regulatory authority for not considering certain atmospheric variables (e.g., change in global O_2 - see Section 6.1, 6.4)
3. Management variables
   - there may be none, unless controls on atmospheric acidity can be instituted
4. Engineering variables
   1. Rainfall quantity (practical)
   2. Material depositional quantities (practical)
   3. Solar radiation fluxes (practical)

VIII Atmospheric Interactions with Drainage Basin-Lake System

1. Description
   - future changes in atmospheric deposition on watershed export to lake
   - future stability of lake levels and vertical mixing patterns
2. Probable geochemical/biochemical mechanisms
   - in watershed, weathering reactions and their modification by terrestrial ecosystem
   - in lake, algal and bacterial regimes (see Section 2.1.3) responding to water chemistry
3. Probable transport mechanism
   - surface runoff will carry soluble and particulate materials; sheet erosion and erosion of bluffs will be negligible
   - little change in depositional features because lake is quite deep, sheltered, and tailings volume is less than 5% of lake volume

Table 8.3: Continued

4. Suggested analysis approach
   - mass balance analysis on major ions to give estimated lake concentration
   - use nutrient input-lake biomass relationships to estimate biota and settling characteristics
   - use drainage basin export characteristics to estimate lake humics and inorganics and their settling characteristics
   - determine sequestering ability of lake particles
   - determine change in all of above due to change in atmosphere characteristics over time
   - determine background concentration of contaminants of interest (e.g., SO_4^{2-}, Ca, TIC, pH Ra, Th,U)
   - determine additional contribution to background due to tailings leachate
   - determine bioconcentration factors and use as basis for dosimetry calculations

5. Management variables
   1. Background concentrations of major ions (practical)
   2. Background concentrations of contaminants(practical)
   3. Marginal increase of background due to tailings leachate(partially practical)

6. Engineering variables
   1. Elemental budgets for lake water and change over time (practical)
   2. Removal rates of soluble elements due to sequestering by particles (partially research)
   3. Bleed rate of tailings contaminants from sediment to water column (practical)

IX Mitigating Measures

1. Description
   - insertion of tailings barrier (clay) has been assumed
   - insertion of salt layers is to be analyzed
2. Probable biochemical/geochemical mechanisms
   - aerobic conditions (Eh = +300 to +400) in fresh water, anaerobic conditions in salty layers
   - pH will be >6.5-6.8 initially and become more acidic if atmospheric acidity continues; pH of salty layers may be lower due to anaerobic fermentation

Table 8.3: Continued

- algal regimes typical of present lakes may exist in surface waters but change if the lake acidifies
- sequestering will occur in the epilimnion (aerobic adsorption will probably dominate) and in the hypolimnion (anaerobic adsorption; and some coprecipitation such minerals as vivanite may occur if sufficient Fe and S are present)

3. Probable transport mechanisms
   - fresh surface waters will be well-mixed annually; bottom salty layers will be dominated by diffusion
   - across the chemocline, diffusion plus chemocline entrainment by surface currents into epilimnion will occur. At some point in the future, complete mixing top to bottom may occur, if sufficient erosion of chemocline has occurred. Provision for inflow-outflow to maintain fresh water will lengthen the time before overturn occurs by maintaining larger density differences (e.g., the stability of the Dead Sea has been maintained until 1979 by fresh water inflows)
   - particles will settle out to lake sediments
   - depositional character of sediments will be increased
4. Suggested analysis approach
   1. Examine existing pathways and biogeochemical mechanisms and decide on changes due to meromixis
   2. Assess stability or rate of breakdown (a useful approach is to use a mathematical model and bulk lake stability numbers - see Section 2.1.1, 6.3, 6.4, 7.2)
   3. Determine the change in transport due to meromixis and the degree of control of bleed rate afforded by the creation of meromixis
5. Management variables
   1. Rate of breakdown of meromixis (practical)
   2. Alteration in bleed rate reaching surface waters (mainly practical)
6. Engineering variables
   1. Alteration in vertical transport rate (use a temperature model to assess; practical)
   2. Alteration in biochemical regimes (practical)
   3. Alteration in transport to surface waters (fairly practical)

9.0 RECOMMENDATIONS

1. Further development of engineering and management parameters is needed. Various management parameters are available for assessing dominant processes (e.g., ratio of advective transport to dispersive transport) and lake stability. Other parameters will evolve. Using existing knowledge, development of other parameters would be aided by a literature study to determine relevant combinations of variables. In particular, variables which combine biochemical and geochemical mechanisms into attenuation coefficients related to transport physics (advection, diffusion) would be useful for engineering and management purposes. Their change over time would need to be related to the changes in biochemical and geochemical systems over time. A distribution coefficient is analogous to an attenuation coefficient, except that its change over time needs to be related to the fundamental mechanisms of uptake and dissolution. Such parameters may allow the setting of specific criteria or provide engineering designers with preliminary estimates for screening of the dominant processes and lake characteristics in a variety of lakes.

2. A generic study of a probable tailings mass located in a variety of lake systems could be undertaken. Major biochemical and geochemical systems would be hypothesized. A mass balance analysis would be conducted; available literature data would be used to describe such uptake-release mechanisms as precipitation-dissolution, adsorption-desorption, and chemical complexation. For consideration of particulates, five types could be considered: clays, granitic particles, algal detritus, humic substances, and chemical precipitates. An error analysis would be made to examine unknown factors and their importance to the estimation of long-term transport. Such a study would aid in the development of engineering and management parameters as described in recommendation 1.

3. Precise knowledge of the chemical form of various contaminants in the tailings before deposition and their leachability under various Eh and pH conditions is needed in order to assess their long-term behaviour. Such variables are probably not routinely analyzed. Knowledge of the crystal structure of precipitates such as $RaBaSO_4$, their long-term diagensis, and the effects of diagensis upon the solubility product of the precipitate is needed.

4. Studies of the effects of anaerobic decay of organic materials on dissolution of sulfide and other accessory minerals is needed.

5. Systematic analysis of adsorption characteristics of tailings contaminants onto various surfaces would be useful.

6. Pure solution chemical studies on the coprecipitation of radionuclides and other potential contaminants with various minerals formed under diagenetic processes (e.g., vivianite) are needed. Valuable information may be obtainable from a clear enunciation of the association of radioisotopes in the host material with various minerals. Information on the minerals which will form under various pore water conditions is much more readily available in a relative sense.

7. Studies which increase man's confidence in his ability to estimate the erosional-depositional character of lakes would be useful. A strong need exists for studies which bracket the effects of sediment erosion upon water quality. Such studies would include estimates of the "hang-time" of particles after erosion and the amount of contaminant released during this state of suspension.

8. Causes and natural controls of meromixis need to be further explored in order to provide design tools for screening candidate lakes proposing this mechanism as a retaining barrier.

9. Mathematical models which are capable of predicting the breakdown of meromixis exist, but may need further development before application to the analysis of tailings management. A main need is to test their predictions of lake physics agains observations on existing meromictic lakes.

10. Theories which provide a general approach for estimating radionuclide and other potential contaminant bioconcentration behaviour will be useful for management purposes, but are only now evolving. Advances in these areas would greatly increase a decision maker's and the public's confidence in dosimetry and toxicity calculations. Data relevant to PreCambrian shield water chemistry are conspicuously lacking.

11. Studies involving test boring in sediments of lake systems subjected to tailings discharge in the past (e.g., Lake Athabaska, lakes in Bancroft area), chemical partitioning of radioisotopes and other ions into soluble and particulate forms within the stabilized tailing mass, and analysis of the mineralogy should be made to check whether existing knowledge of short-term biochemical and geochemical mechanisms explains these observations. Such lake systems span the breadth of possible lakes being considered for tailings disposal (e.g., Athabaska - a wind swept lake with granular sediments; Bancroft lakes - head water lakes with organic sediments).

12. Studies on limnocorrals or laboratory columns could be undertaken to confirm the dominant biochemical and geochemical mechanisms which will exist for a variety of Eh and pH conditions.

13. A critical study which assesses the groundwater near existing ore bodies would allow a determination of the rate of migration of contaminants from the host phase under a given set of Eh and biochemical conditions. For example, peripheral observations of groundwater at some Northern Saskatchewan sites suggest that little migration occurs. The question of concern is what are the differences between the host ore and tailings placed in lake sediments and what inferences can be drawn from such knowledge?

14. The future chemistry of the atmosphere needs assessment. Climatic change is important, but probably relatively unassessable. It is suggested that scenario-testing studies be made to determine the effects of possible climatic changes and atmospheric extrema upon model watersheds. It is suggested that future atmospheric acidity be studied by considering future fuel consumption patterns and atmospheric compensating mechanisms.

LITERATURE CITED

Agemian and A.Y.S. Chau. 1977. A study of different analytical extraction methods for non-detrital heavy metals in aquatic sediments. Arch. Environ. Contam. Toxicol. 6: 69-82.

Allin, J.T. 1978. Heavy metals in the environment: sources and sinks. Can. Soc. Env. Biol. Newsletter/Bulletin, 35(2): 10-17.

Amiard, J.C., and C. Amiard-Triquet. 1980. Le transfert des polluants dans les chaînes alimentaires aquatiques. Ann. Biol, 19: 117-146.

Amiard-Triquet, C. 1979. Synthèse biblographique sur al radioécologie du Plutonium et des autres transuraniens dans les milieux aquatiques. Bull. Ecol, 10: 241-277.

Anderson, T.B., E.C. Tsigolov, and S.D. Shearer. 1961. Effects of uranium mill wastes on biological fauna of the Animas River (Colorado-New Mexico). 1st Nat. Symp. on Radioecol., Ft. Collins, Colo.

Army Engineer Waterways Experiment Station, Vicksburg, MS. 1976. Ecological evaluation of proposed discharge of dredged or fill material into navigable waters: Interim guidance for implementation of Section 404(b)(1) of Public Law 92-500 (Federal Water Pollution Control Act Amendments of 1972. Rep. No. WES-MP-D-76-17. 89p.

Baccini, P., J. Ruchti, O. Wanner, and E. Grieder. 1979. Regulation of trace metal concentrations in limno-corrals. Swiss J. Hydrol, 41: 202-227.

Baker, R.A.(ed.). 1980. Contaminants and Sediments. Vol. 1 and 2. Ann Arbor Science. 558 pp. and 627 pp., respectively.

Barica, J., M.P. Stainton, and A.L. Hamilton. 1973. Mobilization of some metals in water and animal tissue by NTA, EDTA, and TPP. Water Research, 7(12): 1791-1804.

Beak, T.W. 1964. A biotic index of polluted streams and its relationship to fisheries. Second Internat. Conf. on Water Poll. Res., Tokyo, Japan.

BEAK. 1980a. Long term radioactive materials concentrations in a lake overlying mine tailings subject to alternative management schemes. Report submitted to Canada Wide Mines. 19 pp. with figures.

BEAK. 1980b. An assessment of the radiological impacts of uranium mining in Northern Saskatchewan. Report for Environment Canada, August, 180 pp.

Berner, R.A. 1974. Kinetic models for the early digenesis of nitrogen, sulfur, phosphorous and silicon in anoxic marine sediments. In The Sea, Vol. 5, ed. E.D. Goldberg, J. Wiley & Sons, N.Y.

Blanton, J.O. 1973. Vertical entrainment into the epilimnia of a stratified lake. Limnol. Oceanogr., 18: 697-704.

_____. 1973. Rates of vertical entrainment in stratified lakes. International Association of Hydrological Sciences, Pub. No. 109: 301-305.

Bobba, A.G., Bukatarp. 1980. Parametrization of contaminant transport through an aristotrophic unconfined aquifer. Nordic Hydrology, 189-200.

Boddington, M.J., A.S.W. Defreitas, and D.R. Miller. 1979. The effect of benthic invertebrates on the clearance of mercury from sediments. Ecotoxicol. and Environ. Safety, 3(3): 236-244.

Bolshakov, A.P. 1969. Psilomelane from the Nikitovka mercury deposit. Dopov. Akad. Nauk Ukr. R.S.R. Ser. B., Vol. 31, p. 106 (as cited by Chemical Abstracts: CA71-5324j).

Bordovskiy, O.K. 1965. Accumulation and transformation of organic substances in marine sediments. Marine Geol, 3: 1-14.

Bourg, A.C.M. and R.H. Filby. 1974. Adsorption isotherms for the uptake of Zn (2+) by clay minerals in a freshwater medium. Proc. Intl. Conf. on Transp. Persist. Chem. in Aquat. Ecosyst., P. II.1-II.5.

Bourg, A.C.M. 1979. Effect of ligands at the solid-solution interface upon the speciation of heavy metals in aquatic systems. Proc. Int. Conf. on Man & Contr. of Heavy Metals in the Envt. London, Sept. 1979: 446-449.

Brillouin, L. 1962. Science and Information Theory. Academic Press, New York. 347 p.

Brooks, J.L. 1969. Eutrophication and changes in the composition of the Great Lakes. p. 236-255. In Eutrophication: Causes, Consequences, Correctives. Nat. Acad. Sci., Washington, D.C.

Bundi, T. 1979. Ph.D. thesis, Swiss Fed. Inst. Technol.

Carignan, R., and J. Kalff. 1980. Phosphorus sources for aquatic weeds: water or sediments? Science, 207: 987-989.

Carter, J.C.H. 1961. A preliminary ecological study of a meromictic lake in northern Labrador. Ph.D. thesis, McGill Univ., Montréal.

CEA. 1979. Co-ordinated research program on the source, distribution, movement and deposition of radium in inlet waterways and aquifers. Prog. Rept. 1978. Commisariat à l'Energie Atomique.

Cedergren, H.R. 1977. Seepage, Drainage and Flow Nets. Wiley. 500 p.

Champ, D.R., J. Gubens and R.E. Jackson. 1979. Oxidation-reduction systems in groundwater flow systems. Can. J. Earth Sci. 16: 16-23.

Cherry, D.S., S.R. Larrick, R.K. Guthrie, E.M. Davis, and F.F. Sherberger. 1979. Recovery of invertebrate and vertebrate populations in a coal ash stressed drainage system. J. Fish. Res. Bd. Can, 36: 10:61.

Cherry, J.A., R.J. Blackport, N. Dubrovsky, R.W. Gillham, T.P. Lim, D. Murray, E.J. Reardon, and D.J.A. Smyth. 1980. Subsurface hydrology and geochemical evolution of inactive pyritic tailings in the Elliot Lake Uranium District, Canada. Proc. 3rd Sym. on Uranium Tailings Management, C.S.U., Colorado. 32 pp.

Cline, J.T., and G.B. Upchurch. 1975. Proc. 16th Conf. Great Lakes Res.: 349-356.

Cline, J.T., J.B. Hillson, and S.B. Upchurch. 1973. Mercury mobilization as an organic complex. Proc. 16th Conf. Great Lakes Res., 1973: p. 233-242.

Dacey, J.W.H. 1980. Internal winds in water lilies: an adaptation for life in anaerobic sediments. Science. 210, 1017-1019.

Davis, R.B. 1974. Stratigraphic effects of tubificids in profundal lake sediments. Limnol. Oceanogr., 19: 466-488.

Demayo, A., M.C. Taylor, and S.W. Reeder. 1979. Guidelines for surface water quality Vol. 1. Inorganic chemical substances: Nickel. Inland Wat. Dir., Water Qual. Br., Ottawa, Canada.

Dickman, M., M. Ouellet, and S. Severn. 1981. Comparative meromixis in Canada along a latitudinal gradient. Third Annual Meeting, Can. Chapt. of Int. Soc. Theor. & Appl. Limnol., Montréal.

Dillon, P.J., and F.H. Rigler. 1974. A test of a simple nutrient budget model predicting the phosphorus concentration in lake water. J. Fish. Res. Bd. Can., 31: 1771-1778.

_____. 1975. A simple method for predicting the capacity of a lake for development based on a lake trophic status. J. Fish. Res. Bd. Can., 32: 1519-1531.

DiToro, D.M., and J.P. Connolly. 1980. Mathematical model of water quality in large lakes, Part 2: Lake Erie. Report submitted to U.S. Environmental Protection Agency, 230 pp.

Duthie, H.C., and J.C.H. Carter. 1970. The meromixis of Sunfish Lake, southern Ontario. J. Fish. Res. Bd. Can. 27: 847-856.

Edinger, J.E., D.K. Brady, J.C. Geyer. 1974. Heat exchange and transport in the environment. John Hopkins Univ. Report 125 pp.

Elliott, H.A., and C. Huang. 1980. Adsorption of some copper (II)-amino acid at the solid-solution interface. Effect of ligand and surface hydrophobicity. Environ. Sci. Technol, 14: 87-93.

EPS. 1978. A study of water pollution in the vicinity of the Eldorado Nuclear Limited Beaverlodge Operation in 1966 and 1977. Environmental Protection Service Rept. EPS-5-NW-78-10 (Canada).

Farrah, H., and W.F. Pickering. 1977. Influence of clay-solute interactions on heavy metal ion levels. Water Air Soil Pollut, 8: 189-197.

Fee, E.S. 1980. Important factors for estimating annual phytoplankton production in the Experimental Lakes Area. Can. J. Fish. Aquat. Sci., 37: 513-522.

Feldt, W. 1979. Die Aufnahme und Anreicherung radioactiver Stoffe in aquatische Oranismen. Fachtagung "Radioökologie des Deutschen Atomforums, 2-3 Oct. 1979.

Findenegg, I. 1935. Limnologische Untersuchungen in Kärntner Seengebeite. Ein Beitrag zur Kenntnis des Stoffhaushaltes in Alpenseen. Int. Rev. Gesamten Hydrobiol, 32: 369-423.

_____. 1937. Holomiktische und meromiktischer Seen. Int. Rev. Gesamten Hydrobiol, 35: 586-610.

Fogg, G.G. 1966. Algal Cultures and Phytoplankton Ecology. Univ. of Wisconsin Press, Madison, Wisconsin. 126 pp.

Forstner, U. 1977. Forms and sediment associations of nutrients (C, N and P), pesticides and metals. Trace Metals. In: Shear, H. and A.E.P. Watson (eds.). Proc. Workshop on Fluvial Transp. of Sediment-Associated Nutrients and Contaminants, Kitchener, Ontario. Oct. 1976. Internat. Joint Comm. Research and Advisory Bd.: 219-233.

Förstner, U., and G. Müller. 1976. Heavy metal pollution monitoring by river sediments. Fortschr. Mineral 53: 271-288.

Francisco, D.E., R.A. Mah, and A.C. Rabin. 1973. Acridine orange-epifluorescence technique for counting bacteria in natural waters. Trans. Amer. Microsc. Soc., 92: 416-421.

Freeman, N.G., P.F. Hamblin, T.S. Murthy. 1974. Helmholtz resonance in harbors of the Great Lakes. Proc. 17th Conf. Great Lakes Res., Intern. Assoc. Great Lakes Res.: 399-411.

Freeze, R.A. and J.A. Cherry. 1979. Groundwater. Prentice Hall. 604p.

Frey, D.G. 1969. Evidence for eutrophication from remains of organisms in sediments. In Eutrophication: Causes, Consequences, Correctives. Washington, D.C. Nat. Acad. Sci.: 594-613.

_____. 1974. Palaeolimnology. Mitt. Int. Ver. Limnol. 20: 95-123.

Frost, R.R., and R.A. Griffin. 1977. Effect of pH on adsorption of copper, zinc, and cadmium from landfill leachage by clay minerals. J. Environ. Sci. Health, A12 (4 and 5): 139-156.

Gächter, R., and A. Mares. 1979. Effects of increased heavy metal loads on phytoplankton communities. Swiss J. Hydrol, 41: 228-246.

Gale, N.L., and B.G. Wixson. 1979. Control of heavy metals in lead industry effluents by algae and other aquatic vegetation. In: Management and Control of Heavy Metals in the Environment. Internat. Conf., London, Sept. 1979. CEP Consultants Ltd., Edinburgh, Scotland: 580-583.

Gannon, J.E., and A.M. Beeton. 1979. The decline of the large zooplankter Limnocalanus macrurus Sars. (Copepoda: Calanoida) in Lake Erie. Proc. 14th Conf. Great Lakes Research, Int. Assoc. Great Lakes Res.: 27-38.

Garner, R.J. 1972. Transfer of radioactive materials from the terrestrial environment to animals and man. CRC Press.

Giesy, J.P., Jr., and D. Paine. 1977. Uptake of americium-241 by algae and bacteria. Prog. Water Technol., 9(4): 845-857.

Gliwicz, Z.M. 1969. The food sources of lake zooplankton. Ekol. Pol. (Ser. B), 15: 205-223.

Godbec, H.W., E.L. Compere, D.S. Joy, A.H. Kibbey, J.G. Moore, C.W. Nestor, O.U. Anders, R.M. Neilson. 1980. Application of mass transport theory to leaching of radionuclides from waste solids. Nucl. & Chem. Management, 1: 29-35.

Goldberg, E.D., and M. Koide. 1963. Rates of sediment accumulation in the Indian Ocean. In Earth Science and Meteroritics, ed. J. Geiss and E.D. Goldberg. North-Holland Pub. Co., Amsterdam: 90-102.

Goldman, C.R., D.T. Mason, and B.J.B. Wood. 1972. Comparative study of the limnology of two small lakes on Ross Island, Antarctica. In: G.A. Llano (ed.), Antarctic Terrestrial Biology, Amer. Geophys. Union, Washington, D.C.: 1-50.

Golterman, H.L. (ed.). 1977. Interactions between Sediments and Fresh Water. Junk Publ. 473 pp.

Goodman, R.E. 1980. Introduction to Rock Mechanics. Wiley.

Goulden, C.E. 1969. Temporal changes in diversit. In: G.M. Woodwell and H.H. Smith (eds.), Diversity and Stability in Ecological Systems. Brookhaven Symposia in Biology 22: 96-102.

Gray, D.A., P.B. Greenwood, G. Bisson, C.R. Cratchley, R. Harrison, J.D. Mather, D. Ostle, E.G. Poole, B.J. Taylor, and V.P.L. Willmore. 1976. Disposal of highly-active, solid radioactive wastes into geological formations - relevant geological criteria for the United Kingdom. U.K. Natural Environment Research Council Report 76/12, Queen's Printer, London, England. 7 pp.

Guthrie, R.K., D.S. Cherry, and J.H. Rodgers. 1974. The impact of ash basin effluent on biota in the drainage system. Proc. 7th Mid-Atlantic Indust. Waste Conf. Nov. 12-14, 1974. Drexel Univ. Philadelphia, Pennsylvania: 17-43.

Häkanson, L. 1980. An ecological risk indiex for aquatic pollution control. A sedimentological approach. Water Res., 14, 975-1001.

Hall, T. 1978. Nickel uptake, retention and loss in Daphnia magna. M.Sc. thesis, Dept. of Botany, Univ. of Toronto. 195 p.

Hammer, U.T., R.C. Haynes, J.R. Lawrence, and M.C. Swift. 1978. Meromixis in Waldsea Lake, Saskatchewan. Verh. Internat. Verein. Limnol, 20: 192-200.

Hansen, N.E.O. 1979. Effects of boundary layers on mixing in small lakes. In: Hydrodynamics of Lakes. Graf and Mortimer (eds.). Elsevier. 341-356.

Hawley, J.R. 1975. On land or under water? The tailings disposal problem. Ont. Miln. of Envir. Report, 44 pp.

Haworth, E.Y. 1980. Comparison of continuous phytoplankton records with the diatom stratigraphy in the recent sediments of Blelham Tarn. Limnol. Oceanogr. 25: 1093-1103.

Hem, J.D. 1977. Reactions of metal ions at surfaces of hydrous iron oxide. Geochim. Cosmochim. Acta, 41, 527-538.

Hesslein, R.H., W.S. Broecker, and D.W. Schindler. 1978. The fates of metal radiotracers added to a whole lake: Sediment-water interactions. J. Fish. Res. Bd. Can. In prep.

Hesslein, R.H. 1979 Personal communication.

Hesslein, R., and P. Quay. 1973. Vertical eddy diffusion in the thermocline of a small stratified lake. J. Fish. Res. Bd. Can., 30: 1491-1500.

Hilsenhoff, W.L. 1977. Use of anthropods to evaluate water quality of streams. Dept. Natural Res., Tech. Bull. No. 100, Madison, WI. 15 p.

Holtz, R.D. and W.D. Kovacs. 1981. Geochemical Engineering. Prentice-Hall. 733 p.

Huber, W.C., D.R.F. Harleman, and P.J. Ryan. 1972. Temperature prediction in stratified reservoirs. Proc. Am. Soc. Civ. Engr., J. Hyd. Div., 98: 645-666.

Hutchinson, G.E. 1967. A Treatise on Limnology. II. Introduction to Lake Biology and the Limnoplankton. New York, John Wiley and Sons Inc. 1115 pp.

_____. 1973. A contribution to the limnology of arid regions primarily founded on observations made in the Lahontan Basin. Trans. Conn. Acad. Arts & Sci., 33: 47-132.

Hutchinson, T.C., and J. Fitchko. 1974. Heavy metal concentrations and distributions in river mouth sediments around the Great Lakes. Inst. Environ. Studies and Engineering, Univ. Toronto. In: Proc. Int. Conf. Transp. Persist. Chem. in Aquat. Ecosys., Ottawa, Can. May 1-3, 1974.

Hutchinson, T.C., and P.M. Stokes. 1975. Heavy metal toxicity and algal bioassays. Water Quality Parameters, ASTM STP 573, American Society for Testing and Materials, Phila.: 320-343.

IAEA. 1976. Effects of ionizing radiation on aquatic organisms and ecosystems. International Atomic Energy Agency Tech. Rept. Ser. #172.

Imboden, D.M. 1975. Interstitial transport of solutes in non-steady state accumulating and compacting sediments. Earth. Planet. Sci. Letters 27: 221-228.

Inoue, T., and R. Suemitsu. 1979. On a peptide as the bonding material of mercury in the bed sediment. Nippon Nogeikagaku Kaishi, 53(2): 45-50.

Jackson, R.E., and K.J. Inch. 1980. Hydrogeochemical processes affecting the migration of radionuclides in a fluvial sand aquifer at the Chalk River nuclear laboratories. Nat. Hydrol. Res. Inst. Paper No. 7, Sci. Ser. No. 104. Inland Waters Directorate, Ottawa.

Jackson, T.A. 1975. Humic matter in natural waters and sediments. Soil Sci. 119: 56-64.

_____. 1977. Immobilization of toxic metal pollutants in inland waters by production of algal blooms and hydrogen sulfide. Proc. 20th Limnological Congress, Copenhagen; 7 Aug. 1977.

_____. 1978. The biogeochemistry of heavy metals in polluted lakes and streams at Flin Flon, Canada, and a proposed method for limiting heavy metal pollution of natural waters. Environ. Geol, 2(3): 173-189.

Jackson, T.A., G. Kipphut, R.H. Hesslein, and D.W. Schindler. 1980. Experimental study of trace metal chemistry in soft-water lakes at different pH levels. Can. J. Fish. Aquat. Sci, 37: 387-402.

Jenne, E.A. 1968. Controls on Mn, Fe, Co, Ni, Cu and Zn concentrations in soils and water: The significance of hydrous Mn and Fe Oxides. In: Trace Inorganics in Water. Am. Chem. Soc. Adv. Chem. Ser. No. 73, 337-387.

Jenne, E.A., and S.N. Luoma. 1977. Forms of trace elements in soils, sediments and associated waters: An overview of their determination and biological availability. In: Proc. Symp. of Biol. Implc. of Metals in the Envt. ERDA Symp. Series 42: 110-143.

Jonasson, I. 1977. Geochemistry of sediment/water interactions of nutrients, pesticides and metals, including observations on availability. Metals. In: Shear, H., and A.E.P. Watson (eds.) Proc. Workshop on the Fluvial Transport of Sediment-Associated Nutrients and Contaminants. Kitchener, Ont. Oct. 1976. Internat. Joint Comm. Research Advisory Board: 255-271.

Kaushik, N.K. and H.B.N. Hynes. 1971. The fate of dead leaves that fall into streams. Arch. Hydrobiol., 68(4): 465-515.

Kaesler, R.L., and E.E. Herricks. 1977. Analysis of data from biological surveys of streams: diversity and sample size. Water Res. Bull., 13 (1): 125-135.

Keely, J.E., and R.M. Engler. 1974. Dredged material research program - Misc. Paper D-74-14. U.S. Army Engineer Waterways Expt. Stat., Vicksburg, MS.

Kemp, P.H. 1971. Chemistry of natural waters - IV Saturation pH value. Water Res. 5: 735-740.

Kenaga, E.E. 1980. Correlation of bioconcentration factors of chemicals in aquatic and terrestrial organisms with their physical and chemical properties. Env. Sci. Technol, 14: 553-555.

Kennedy, E.J., R.R. Ruch, and N.F. Shimp. 1971. Distribution of mercury in unconsolidated sediments from southern Lake Michigan. Studies of Lake Michigan bottom sediments No. 7. Illinois. Geol. Surv. Environ. Geol. Note.

Khalid, R.A., R.P. Gambrell, R.P. Verloo, and M.G. Patrick, Jr. 1977. Transformations of heavy metals and plant nutrients in dredged sediments as affected by oxidation reduction potential and pH, Vol. 1: literature review. U.S. Army Eng., Waterways Exper. Stn., Environ. Effects Lab.

Kilborn. 1980. An assessment of the long term suitability of present and proposed methods for the management of uranium mine tailings report submitted to Atomic Energy Control Board, 250 pp. with 3 appendices.

Kirchner, W.B., and P.J. Dillon. 1975. An empirical method of estimating the retention of phosphorus in lakes. Water Resources Res., 2, 182-183.

Koyanagi, T., M. Nakahara, and M. Imura. 1978. Adsorption of sediment-bound radionuclides through the digestive tract of marine demersal fishes. J. Radiat. Res., 19: 295-305.

Krezoski, J.R., S.C. Mozley, and J. Robbins. 1978. Influence of benthic macroinvertebrates on mixing of profundal sediments in southeastern Lake Michigan. Limnol. Oceanogr., 23(5): 1011-1016.

Kudo, A., and D.C. Mortimer. 1979. Pathways for mercury uptake by fish from bed sediments. Environ. Pollut. 19(3): 239-245.

Kuznetsov, S.I. 1970. Microflora ozer i ee geokhimicheskaya deyate ('nost'). (Microflora of lakes and their geochemical activities). (In Russian). Leningrad, Izdatel'stvo Nauka, 440 pp.

Lahar, N., and M. Hochberg. 1975. Fixation of iron and zinc applied as chelates into a soil column during leaching. Soil Sci. Soc. Amer. Proc., 39: 1213.

Landa, E. 1980. Isolation of uranium mill tailings and their component radionuclides from the bioshphere - some earth science perspectives. U.S.G.S. Circular 814, 32 pp.

Langelier, W.F. 1946. Chemical equilibria in water treatment. J. Am. Water Works Assoc. 38: 169.

Laube, V., S. Ramamoorthy, and D.J. Kushner. 1979. Mobilization and accumulation of sediment bound heavy metals by algae. Bull. Environ. Contam. Toxicol., 21(6): 763-770.

Lee, G.F., R.A. Jones, F.Y. Saleh, G.M. Mariani, and D.H. Homer. 1978. Evaluation of the elutriate tests as a method of predicting contaminant release during open water disposal of dredged sediments and environmental impact of open water dredged material disposal; Volume II. Data Report. N.T.I.S., Springfield, VA. 22161, Rept. No. AD-A-61 710. 1234 p.

Leppard, G.G., A. Massalski, and D.R.S. Lean. 1977. Electron-opaque microfibrils in lakes: their demonstration, their biological derivation and their potential significance in the redistribution of cations. Protoplasma, 92: 289-309.

Lerman, A., and M. Stiller. 1969. Vertical eddy diffusion in Lake Tiberias. Verh. Int. Verein. Limnol., 17: 323-333.

Lerman, A., and H. Taniguchi. 1972. Strontium 90 - Diffusional transport in sediments of the Great Lakes. J. Geophic. Res., 77(3), p. 474.

Li, Y.H. 1973. Vertical eddy diffusion coefficient in Lake Zurich. Schwiez. Z. Hydrol., 35: 1-7.

Likens, G.E., P.H. Bormann, R.S. Pierce, J.S. Eaton, and N.M. Johnson. 1977. Biogeochemistry of a forested ecosystem. Springer-Verlag. 146 pp.

Liklejma, L. 1979. Personal communication. Canada Centre for Inland Waters.

Lovelock, J.E. 1979. GAIA. Oxford University Press. 157 pp.

Mantoura, R.F.C., A. Dickson, and J.P. Riley. 1978. The complexation of metals with humic materials in natural waters. Estuar. Coastal & Mar. Sci., 6: 387-408.

Mayes, R.A., A.W. McIntosh, and V.L. Anderson. 1977. Uptake of cadmium and lead by a rooted aquatic macrophyte (Elodea canadensis). Ecology, 58(5): 1176-1180.

McGaughey, L.M. and J.V. Matson. 1980. Prediction of the calcium carbonate saturation pH in cooling water. Water Res. 14: 1729-1735.

McLaren, I. 1967. Physical and chemical characteristics of Ogac Lake, a land-locked fjord on Baffin Island. J. Fish. Res. Bd. Can., 24: 981-1015.

Michard, G. 1975. Chemistry of Mn at the water-sediment interface. Application to the formation of Mn deposits in ocean sediments. Mineralium Deposita Berlin, 10: 315.

Moore, J.W. 1979. Diversity and indicator species as measures of water pollution in a subarctic lake. Hydrobiologia, 66(1): 840-847.

Morel, F.M., J.C. Westall, C.R. O'Melia, and J.J. Morgan. 1975. Environ. Sci. Technol., 9, 756.

Mortimer, C.H. 1942. The exchange of dissolved substances between mud and water in lakes, III and IV. J. Ecol., 30: 147-201.

_____. 1969. Physical factors with bearing on eutrophication in lakes in general and in large lakes in particular. In: Eutrophication: Causes, Consequences, Correctives. Nat'l. Acad. of Sci., Washington, D.C.: 340-368.

Mortimer, C.H. 1979. Strategies for coupling data collection and analysis with dynamic modelling of lake motions. In: Hydrodynamics of Lakes. Graf & Mortimer (eds.). Elsevier, 183-222.

Mortimer, D.C. 1977. Mercury uptake by aquatic plants; laboratory and field studies. Dist. and Transp. of Pollutants in Flowing Wat. Ecosy., 2:22.1-2.38.

Mortimer, D.C., and A. Kudo. 1975. Interaction between aquatic plants and bed sediments in mercury uptake from flowing water. J. Environ. Qual, 4: 491.

_____. 1977. Interactions between aquatic plants and sediments, invertebrates, and fish with respect to transport and accumulation of mercury. Dist. Transp. Pollut. in Flow. Wat. Ecosys., 2: 23.1-23.22.

Mudroch, A., and J.A. Capobianco. 1979. Effects of mine effluents on the uptake of Co, Ni, Cu, As, Zn, Cd, Cr and Pb by aquatic macrophytes. Hydrobiologia, 64(3): 223-231.

Neuwirth, F. 1979. Experiences from the estimation of evaporation from a shallow lake. In: Hydrodynamics of Lakes. Graf & Mortimer (eds.). Elsevier, 313-325.

Nieboer, E., and D.H.S. Richardson. 1980. The replacement of the non-descript term "heavy metals" by a biologically and chemically significant classification of metal ions. Env. Poll. (Series B), 1: 3-26.

Northcote, T.G., and W.E. Johnson. 1964. Occurrence and distribution of sea water in Sakinaw Lake, British Columbia. J. Fish. Res. Bd. Can., 21: 1321-1324.

Northcote, T.G., M.S. Wilson, and D.R. Hurn. 1964. Some characteristics of Nitinal Lake, an inlet on Vancouver Island, British Columbia. J. Fish. Res. Bd. Can., 21: 1069-1081.

Nriagu, J.O. 1972. Stability of vivianite and ion-pair formation in the system $Fe_3(PO_4)_2 - H_3PO_4 - H_2O$. Geochim. Cosmochim Act. 36, p. 459.

Nriagu, J.O., and C.I. Dell. 1974. Diagenetic formation of iron phosphates on recent lake sediments. American Mineralogist, 59, 934-946.

Nygaard, G. 1955. On the productivity of five Danish waters. Verh. Int. Ver. Limnol. 12: 123-133.

Occhiogrosso, T.J., W.T. Waller, and G.J. Lauer. 1979. Effects of heavy metals on benthic macroinvertebrate densities in Foundry Cove on the Hudson River. Bull. Environ. Contam. Toxicol, 22(1-2): 230-237.

Octavio, K.H., G.H. Jirka, and D.R.F. Harleman. 1977. Vertical heat transport mechanisms in lakes and reservoirs. MIT, Dept. of Civil Engr., Tech. Report #227, Aug. 1977.

Olszewski, P. 1951. The thermal conditions of mountain lakes. Schweiz. Z. Hydrol., 34: 1-33.

O'Melia, C.R. 1972. An approach to the modelling of lakes. Schweiz. Z. Hydrol., 34: 1-33.

Ongley, E.D. 1978. Sediment-related phosphorus, trace elements and anion flux in two rural southeastern Ontario catchments. (Unpublished data, as cited by Allan 1979).

Onishi, Y., D.L. Schreiber, and R.B. Codell. 1980. Mathematical simulation of sediment and radionuclide transport in the Clinch River, Tennessee. In: Contaminants and Sediments Volume 1 (R.A. Baker, ed.). p. 393-406.

Orlob, G.T. 1975. Present problems and future prospects of ecological modeling. In: Ecological Modeling, C.S. Russel (ed.), Washington, D.C., Resources for the Future Inc., 283-313.

Ouellet, M. 1979. Heavy metal distribution and grain size of the superficial sediments of Lake St. John, Quebec. Manage. Control Heavy Met. Environ., Intl. Conf. 1979, Edinburgh, Scotland: 501-504.

OWRC. 1971. Water pollution from the uranium mining industry in Elliot Lake and Bancroft areas. Vol. 1, summary. Ontario Water Resources Commission.

Patalas, K. 1971. Crustacean plankton and the eutrophication of the St. Lawrence Great Lakes. J. Fish. Res. Bd. Can. 29: 1451-1462.

Patrick, D.W., Jr. 1977. Trace and toxic metal uptake by marsh plants as affected by Eh, pH and salinity. U.S. Army Engineer Waterways Expt. Station, Environ. Effects Lab., Vicksburg, MS. 39180. Tech. Rept. No. D-77-40. 124 p.

Payne, K., and W.R. Pickering. 1975. Influence of clay-solute interactions on aqueous copper ion levels. Water Air Soil Pollut., 5: 63-69.

Peet, R.K. 1974. The measurement of species diversity. Ann. Rev. Ecol. Systematics, 5: 285-307.

Pentreath, R.J. 1979. The roles of food and water in the accumulation of radionuclides by marine teleost and elasmobranch fish. In: Radioactive contamination of the marine environment: p. 421-426. International Atomic Energy Agency Report. Vienna.

Pielou, E.C. 1975. Ecological diversity. Wiley Interscience, New York. 165 p.

Plate, E.J., and P. Wengefield. 1979. The exchange processes at the water surface. In: Hydrodynamics of lakes. Graf & Mortimer (eds.). Elsevier, 277-302.

Powell, T., and A. Jassby. 1974. The estimation of vertical eddy diffusivities below the thermocline in lakes. Water Resour. Res., 10: 191-198.

Pradel, J., and P. Zettwoog. 1977. Contamination radioactive des sites miniers au cours des operations d'extraction et de traitement du minéral. Commisariat de l'Energie Atomique.

Priestly, C.H.B., and R.J. Taylor. 1972. On the assessment of surface heat flux and evaporation using large-scale parameters. Monthly Weather Review, 100, (#2), 61-92.

Prosi, F. 1979. Bioavailability of heavy metals in different freshwater sediments: uptake in macrobenthos and biomobilization. In: Management and Control of Heavy Metals in the Environment. Internat. Conf. London, Sept. 1979. CEP Consultants Ltd., Edinburgh, Scotland: 288-291.

Ramamoorthy, S., S. Springthorpe, and S. Kushner. 1977. Competition for mercury between river sediment and bacteria. Bull. Environ. Contam. Toxicol., 17(5): 505-511.

Reckhow, K.H. 1978. Empirical lake models for phosphorus: Development, applications, limitations and uncertainty. In: Perspectives on lake ecosystem modeling, D. Scavia and A. Robertson (eds.). Ann Arbor Sci., Ann Arbor, Michigan. 193-221.

Reinhard, D., and U. Foerstner. 1975. Metal accumulation in sediment cores from locks in the Middle Neckar River. N. Jb. Geol. Palaeont. Mh., 5: 301-320.

Schindler, D.W., and S.K. Holmgren. 1971. Primary production and phytoplankton in the Experimental Lakes Area, northwestern Ontario, and other low carbonate waters, and a liquid scintillation method for determining ^{14}C activity in photosynthesis. J. Fish. Res. Bd. Can., 28: 189-201.

Schindler, D.W, R.H. Hesselin, R. Wageman and W.S. Broecker. 1980. Effects on acidification on mobilization of heavy metals and radionuclides from the sediments of a freshwater lake. Can. J. Fish. Aquat. Sci., 37: 373-377.

Schnitzer, M. 1971. Metal-organic matter interactions in soils and waters. In: Organic Compounds in Aquatic Environments. Faust, S.D. and J.V. Hunter (eds.). Marcel Dekker, Inc., New York.

Schnoor, J.L., and D.J. O'Connor. 1980. A steady state eutrophication model for lakes. Water Res., 14, 1651-1665.

Shannon, C.E., and W. Weaver. 1949. The mathematical theory of communication. Univ. Illinois Press, Urgana, Illinois. 125 p.

Snodgrass, W.J. 1977. Relationship of vertical transport across the thermocline to oxygen and phosphorus regimes: Lake Ontario as a prototype. In: Transport processes in lakes and oceans (Gibbs and Shaw, eds.), Plenum Press, 1979-202.

_____. 1978. Predictive water quality models for the Great Lakes: Some capabilities and limits. In: Perspectives on Lake Ecosystem Modeling. D. Scavia and A. Robertson (eds.). Ann Arbor Sci., Ann Arbor, Michigan, 171-191.

_____. 1980. Distribution and behaviour of nickel in the aquatic environment. In: Nickel in the Environment, J.O. Nriagu (ed.). Wiley, 203-274.

Snodgrass, W.J., and C.R. O'Melia. 1975. A predictive model for phosphorus in lakes. Environ. Sci. and Tech., 10: 937-944.

Spalding, D.B., and U. Svensson. 1976. The development and erosion of the thermocline. Imperial College of Science and Technology, London, England. Paper present at 1976 3 CHM meeting, Dubrovnic, 10 pp.

Stahl, J.B. 1959. The developmental history of the chironomid and Chaoborus faunas of Myers Lake. Invest. Indiana Lakes and Streams 5: 47-102.

Stauffer, R.E., and G.F. Lee. 1973. The role of thermocline migration in regulating algae blooms. In: Modelling the Eutrophication Process. Middlebrooks, E.J., et al (eds.). Proceedings of Workshop, Div. of Envir. Engr., Utah State Univ., Logan, Utah: 73-82.

Stumm, W. 1966. Redox potential as an environmental parameter; conceptual significance and operational limitation. 3rd Internat. Conft. on Water Poll. Res., Water Poll. Contr. Fed. Sect. 1, Paper 13, 16 pp.

Stumm, W., and J.J. Morgan. 1970. Aquatic Chemistry. Wiley Interscience, New York. 583 p.

Sweers, H.E. 1969a. Structure, dynamics and chemistry of Lake Ontario. Manuscript Report Series No. 10, Mar. Scit. Bi., Dept. of Energy, Mines and Resources, Ottawa, Canada.

_____. 1969b. Removal of contaminants from Lake Ontario by natural processes. Proc. 12th Conf. Great Lakes Res., Inter. Assoc. Great Lakes Res.: 734-741.

_____. 1970. Vertical diffusivity coefficient in a thermocline. Limnol. Oceanogr., 15: 273-280.

Szalay, A. 1969. Cation exchange properties of humic acids and their importance in the geological enrichment of UO_2^{++} and other cations. Geochim. et Cosmochim. Acta., 28: 1605-1614.

Terzaghi, K. and R.P. Peck. 167. Soil Mechanics in Engineering Practice. Wiley. 729 p.

Tessenow, U., and Y. Baynes. 1975. Redox-dependent accumulation of Fe and Mn in a littoral sediment-supporting Isoetes lacustris L. Naturwissenschaften, 62: 342-343.

Thomann, R.V., D.M. DiToro, R. Winfield, and D.J. O'Connor. 1975. Mathematical modeling of phytoplankton in Lake Ontario. 1. Model development and verification. U.S. EPA 600/3-75-005, Corvallis, Oregon. 177 p.

Thompson, S.E., C.A. Burton, D.J. Quinn and Y.C. Ng. 1972. Concentration factors of chemical elements in edible aquatic organisms. US AEC Rept. UCRL-50564. Lawrence Radiation Laboratory.

Thunmark, S. 1945. Zur soziologie des Süsswasserplanktons. Eine methodologisch-ökologische studie. Folia Limnol. Scanin. 3, 66 p.

Townsend, D.R., A. Kudo, H. Sayeed, and D.R. Miller. 1974. Mercury transport by bed sediment movements. Proc. Intl. Conf. on Transport of Persistent Chem. ion Aquatic Ecosystems: 47-51.

Troup, B.N. 1974. The interaction of iron with phosphate, carbonate and sulphide in Chesapeake Bay interstitial waters: thermodynamic interpretation. Ph.D. dissertation. John Hopkins Univ., Baltimore, MD.

Turekian, K.K. 1977. The fate of metals in oceans. Geochim. Cosmochim. Acta. 41, 1139-1144.

Turner, J.S., and E.B. Krauss. 1967. A one-dimensional model of the seasonal thermocline. Tellus, 19: 88-105.

United States Environmental Protection Agency/United States Corps of Engineers Techical Committee on Criteria for Dredged and Fill Material. 1977. Ecological evaluations of proposed discharge and dredged material into ocean waters. Implementation manual for Section 103 of Public Law 92-532. U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

United States Nuclear Regulatory Commission. 1977. Calculation of annual doses to man from routine releases of reactor effluents for the purpose of calculating compliance with 10 CRF50, Appendix I. U.S. NRC Guide 1.109.

van der Berg, C.M.G., P.T.S. Wong, and Y.K. Chan. 1979. Measurement of complexing materials excreted from algae and their ability to ameliorate copper toxicity. J. Fish. Res. Bd. Can., 36, 901-905.

Walker, K.F., and G.E. Likens. 1975. Meromixis and a reconsidered typology of lake circulation patterns. Verh. Internat. Verein. Limnol., 19: 442-458.

Weber, W.J., Jr. 1972. Physiochemical Processes for Water Quality Control. Wiley Interscience, New York, 640 p.

Wells, L. 1970. Effects of alewife predation on zooplankton populations in Lake Michigan. Limnol Oceanogr. 15: 556-565.

Welsh, P., and P. Denny. 1976. Waterplants and the recycling of heavy metals in an English lake. In: Hemphill, D.D. (ed.), Proc. Univ. Missouri 10th Annual Conf. on Trace Metals in Environmental Health: 217-223.

Wentsel, R., A. McIntosh, and N. Anderson. 1977. Sediment contamination and benthic macroinvertebrate distribution in a metal-impacted lake. Environ. Pollut., 14: 187-193.

Wetzel, R.G. 1975. Limnology. W.B. Saunders, Philadelphia. 743 p.

Whittaker, R.H. 1972. Evolution of measurement of species diversity. Taxon., 21: 213-251.

Whilhm, J.L. 1967. Comparison of some diversity indices applied to populations of benthic macroinvertebrates in a stream receiving organic wastes. J. Wat. Poll. Contr. Fed., 39(10): 1673-1683.

Williams, P.M., W.H. Mathews, and G.L. Pickard. 1961. A lake in British Columbia containing old sea water. Nature., 191: 830-832.

Woodiwiss, F.S. 1964. The biological system of stream classification used by the Trent River Board. Chemistry and Industry: 443-447.

Yan, N.D., and R. Strus. 1980. Crustacean zooplankton communities of acidic, metal-contaminated lakes near Sudbury, Ontario. Can. J. Fish. Aquat. Sci. 37: 2282-2293.

Yeasted, J.G., and F.M. Morel. 1977. In: Fate of pollutants in the air and water environments. Suffet (ed.). Wiley, Vol. 8, Part 1.

GLOSSARY OF SELECTED TERMS

Aerobic water	water containing oxygen, used by biota for respiration.
Adsorption coefficient	the number relating the mass of a material adsorbed in the particulate phase to the mass in solution.
Advection	movement of a fluid in a particular direction.
Amorphous	pertaining to a solid which is non-crystalline, having neither definite form nor structure.
Anaerobic water	water devoid of oxygen in which materials other than nitrate and oxygen (e.g. sulfate) is the final electron acceptor for respiration.
Anion	an ion that is negatively charged or a radical which moves to the positive pole (anode) in an electric field.
Anoxic water	water devoid of oxygen but containing nitrate which is used as the final electron acceptor for bacterial respiration.
Benthic	pertaining to, or living on the sediments of a body of water.
Biochemistry	the study of chemical substances occurring in living organisms and the reactions and methods for identifying these substances.
Biogenic	produced by actions of living organisms.
Biogeochemistry	a branch of geochemistry that is concerned with biologic materials and their relation to earth chemicals.

Bioaccumulation Factor	the ratio of the concentration of a substance (usually an element) in an organism to the concentration of the same substance on a reference medium (usually water or food).
Bioavailability	that property of a substance (e.g. an element) describing its availability for uptake by organisms, depending upon chemical form.
Bioturbation	the disturbance of sediments by the activity of organisms.
Black box	the description of any component (e.g., a lake) of the biosphere as having a known input and output without knowledge of the component's detailed internal strucutre.
Bleed rate	the rate at which radioisotopes or other contaminants are transported out of tailings.
Blue-green algae	the common name for members of the Cyanophyceae.
Buffering system	the chemical system which controls the pH of a water and prevents a significant change in pH if acid or base is added.
Bulk diffusivity	The average rate of diffusion out of the tailings mass.
Calcite	$CaCO_3$, one the commonest minerals, the principal constituent of limestone; hexagonal-rhombohedral crystal structure, dimorphous with aragonite. Also known as calcopar.
Cation	a positively charged ion or a radical which moves to the negative pole (cathode) in an electric field.
Chemical gradient	the change in chemical concentrations over a given distance.

Chemocline	the transition in a meromicitc lake between the mixolimnion layer (at the top) and the monimolimnion layer (at the bottom).
Consolidation (of sediments)	1. processes by which loose, soft, or liquid earth become coherent and firm. 2. adjustment of a saturated soil in response to increased load; involves squeezing of water from the pores and a decrease in void ratio.
Convection	movement of a fluid in a given direction. Also known as bulk flow.
Crenogenic meromixis	defined on page 198.
Crystalline	of, pertaining to, resembling, or composed of crystals.
Diatom	the common name for algae composing the class Bacillariophyceae; noted for the symmetry and sculpturing of siliceous cell walls.
Diffusion	the spontaneous movement and scattering of particles (atoms and molecules), liquids, gases and solids. At a molecular level, the scattering of a material like arsenic occurs due to it being bombarded by the molecules of the parent fluid.
Diffusivity	a term used to describe the apparent scattering of particles or solute due to eddy processes and molecular diffusion.
Dimictic	defined on page 197.
Dispersion	movement of a solute due to molecular processes, eddy diffusion and advective motion. Most commonly used to describe transport which is mainly in one direction (e.g., a river or groundwater). In surface waters, eddy diffusion and advective motions dominate where as molecular diffusion and advective motions dominate in groundwater.

Dissolved organic carbon (DOC)	organic carbon which is in a dissolved state in water. Often, the term soluble organic carbon is used interchangeably with dissolved organic carbon. The operating definition of dissolved relates to materials which pass through a 0.45 um filter.
Distribution coefficient	the ratio of the amounts of solute dissolved in two immiscible liquids at equilibrium. More commonly defined as ratio of amount of material adsorbed to or incorporated into a particulate phase to that in solution.
Ectogenic meromixis	defined on page 198.
Eddy diffusion	diffusion which occurs in turbulent flow, by the rapid process of mixing in the swirling eddies of fluid. Also known as turbulent diffusion.
Epilimnion	an upper stratum present in lakes during summer, characterized by more or less uniformly warm, circulating, and occasionally turbulent water; separated from the denser, cool bottom layer (hypolimnion) by the metalimnion.
Fulvic acids	complex organic acids, soluble in acids such as sulfuric acid. They may result from the breakdown of humic acids or celluiar materials (e.g., algae) or from the agglomeration of simpler molecules.
Geochemistry	the study of the chemical composition of various phases of the earth and the physical and chemical processes which have produced the observed distribution of the elements in these phases.
Green algae	the common name for members of the plant division Chlorophyta.
Gypsum	$CaSO_4.2H_2O$, a mineral, the commonest sulfate mineral; crystals are monoclinic, clear, white to gray, yellowish, or brownish in colour, with well-developed cleavages.

Humic acid	any of various complex organic acids obtained from humus; insoluble in acids and organic solvents.
Hydraulic gradient	the slope of the hydraulic grade line of a stream; the slope of the free surface of groundwater or the slope of the hydrostatic height of artesian groundwater.
Hypolimnion	a lower stratum in lakes during summer, characterized by cold and relatively undisturbed water, separated from the less dense surface layer (epilimnion) by the metalimnion.
Invertebrate	an animal lacking a backbone and internal skeleton.
Isotactic rebound	the return of a geological structure to an equilibrium position after being subjected to an external pressure such as a glacier.
Jarosite	$KFe_3(SO_4)_2\ (OH)_6$, an ocher-yellow or brown alunite mineral having rhombohedral crystal structure. Jarosite is typically formed during and leaching of U-ores, but may be formed in acidic tailings.
Leaching	the separation or "dissolving out" of soluble constituents from a rock or ore body by percolating water.
Ligand	the molecule, ion, or group bound to the central atom in a chelate or a co-ordination compound.
Macrobenthic organisms	visible, macroscopic organisms living in association with a substrate (such as the sediment) in an aquatic environment.
Macrophyte	a macroscopic plant. In aquatic environments, large filamentous algae and rooted plants (e.g., waterweeds, water lilies, reeds, etc.) are included.

Meromictic	of, or pertaining to a lake whose water is permanently stratified in terms of density and, therefore, does not show complete vertical circulation at any time during the year.
Metalimnion	the stratum of steep, thermal gradient in a lake stratified thermally in the vertical direction.
Mineralization	1. the process of fossilization whereby inorganic materials replace the organic constituents of an organism. 2. the introduction of minerals into a rock, resulting in a mineral deposit.
Mixolimnion	defined on page 197.
Molecular diffusion	the transfer of mass between adjacent layers of fluid in laminar flow; the transport of mass due to bombardment of molecules of a fluid.
Monimolimnion	the dense bottom stratum of a meromictic lake; it is stagnant and does not mix with the water above.
Monomictic	defined on page 197.
Oxic water	water containing oxygen.
Particulate organic carbon (POC)	fraction of organic carbon in water larger in diameter than a standard defined size (typically >0,45 um).
Partition coefficient	ratio of the equilibrium concentrations of a solute in two combined immiscible liquids.
Permeability	the capacity of a porous rock, soil, or sediment for transmitting a fluid without damage to the structure of the medium.
Picnocline	defined on page 197.
Piezometric gradient	the change of the hydrostatic pressure of groundwater or surface water over a distance; the slope of the hydrostatic pressure.

Plate tectonics	global tectonics based on a model of the earth characterized by a small number (10-25) of semi-rigid plates which float on some viscous underlayer in the mantle; each plate moves more or less independently and grinds against the others, concentrating most deformation, volcanism, and seismic activity along the periphery.
Precipitation	the process of producing a separable, solid phase within a liquid medium; the formation of water droplets from water vapour which falls to the earth as rain or ice crystals.
Redox potential	is proportional to the equivalent free energy charge per mole of electrons associated with a given reduction and describes the state of reduction or oxidation in the system; measured in units of millivolts; often expressed as Eh where a correction is made for the change in redox at the existing pH to pH = 7.
Richardson number	a dimensionless number used in the study of stratified flow of multilayer systems; equal to the acceleration of gravity times the density gradient of a fluid, divided by the product of the fluid's density and the square of its velocity gradient at a wall.
Rock matrix	the microscopic composition of parent rock (e.g., sandstone) containing minerals.
Sequestering agent	a substance that removes a metal ion or compound from a solution system by forming a complex ion that has less chemical reactivity than the ion that is removed; can be a chelating or a complexing agent.
Shearing stress	a stress in which the material on one side of a surface pushes on the material on the other side of the surface with a force which is parallel to the surface. Also known as shear stress.

Sink	a device of system where some extensive entity is absorbed, such as a heat sink, a sink flow, a load in an electric circuit, or a region in a nuclear reactor where neutrons are strongly absorbed; a region in the biosphere or earth into which a substance tends to accumulate.
Source	a region in the biosphere (e.g., U-mill tailings) which can supply a material to the biosphere wheter due to purposeful intent or material weathering (e.g., leaching).
Stratification	the arrangement of a body of water into two or more horizontal layers of differing characteristics, especially densities.
Thermocline	a temperature gradient as in a body of water, in which the temperature decrease with depth is greater than that of the overlying and underlying water. Generally defined as the zone where the change in temperature with depth is $\geq 1°C/m$.
Total inorganic carbon (TIC)	all forms of dissolved inorganic carbon, usually the sum of dissolved carbon dioxide, bicarbonate and carbonate.
Vivianite	$Fe_3(PO_4)_2 \cdot 8\ H_2O$, a colourless, blue, or green mineral in the unaltered state, crystalline in the monoclinic system and occurs in earth form and as globular and encrusting fibrous masses.
Wind-driven mixing	hydrodynamic mixing of lakes (or oceans) caused by wind stresses at the surface (as opposed to mixing caused by other forces such as the earth's rotation and river inflows).